UNIVERSITY OF ALBERTA

DISSOLVED PHOSPHORUS IN SHALLOW SUBSURFACE WATER IN AN UNCUT AND CUT SUBCATCHMENT OF A LAKE ON THE BOREAL PLAIN

BY

JEANETTE ELIZABETH EVANS ©

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For God has not given us a spirit of fear,

But of power, and of love,

And of a sound mind.

- 2 Timothy 1:7

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The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitled **Dissolved phosphorus in shallow subsurface water of an uncut and a cut subcatchment of a lake on the Boreal Plain** submitted by **Jeanette Elizabeth Evans** in partial fulfillment of the requirements for the degree of **Masters of Science** in **Environmental Biology and Ecology**.

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To my parents, Dewi and Diana Evans,
for their everlasting support throughout my life

ABSTRACT

Effects of logging on total dissolved phosphorus concentrations ([TDP]) in shallow subsurface water (< 2.5 m deep) were assessed by sampling nested wells and piezometers in an uncut and cut subcatchment of a boreal lake. Wells collected water from mineral and organic soil layers; piezometers collected water from mineral soil layers. [TDP] generally decreased with depth. [TDP] in wells were higher or similar to concentrations in adjacent piezometers. [TDP] in wells were higher when water flowed through the organic layer, reflected in the positive relationship with transmissivity (r^2=0.5), potassium (r^2=0.7) and dissolved organic carbon (r^2=0.5) concentrations. In piezometers, [TDP] were negatively related to calcium (r^2>0.5), magnesium (r^2>0.5), and sodium (r^2=0.7) concentrations. [TDP] in wells and export were higher in the cut subcatchment (P<0.01). However, more water flowed through the organic layer in the cut subcatchment, making it difficult to separate logging effects from differences in water flow between the subcatchments. As logging often increases subsurface flow, logging will likely increase [TDP] in, and export from, areas where the water table is close to the surface.

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Chapter 1 Literature Review

Phosphorus (P) dynamics in forest ecosystems can influence P in adjacent lakes and streams via subsurface flow (water flowing through the ground). P is needed by all living organisms and often limits algae and aquatic plant growth in lakes (Schindler 1977; Prepas and Trimbee 1988). Although P is cycled by living organisms, the P cycle is also influenced by other physiographic characteristics, such as geology, lithology (soils), hydrology, topography, and climate. When trees are harvested, the amount of P transported by water can increase due to decreased P uptake by plants and increased P accumulation and subsurface flow. This literature review focuses on the effects of physiographic characteristics and forest harvesting on P mobility in the forest ecosystem with an emphasis on the boreal mixedwood forest on the Boreal Plain of Canada.

Phosphorus in soil

The underlying geology can contribute various amounts of P to soils mainly due to differences in stability. Igneous rock is not as easily weathered as sedimentary rock and therefore releases less P over a given period of time (Anderson 1988). For example, mean total P (TP) export from streams on sedimentary rock on the Boreal Plain (12.3 kg km^{-2} yr^{-1}, Cooke and Prepas 1998) and on the Precambrian Shield (10.7 kg km^{-2} yr^{-1}, Dillon and Kirchner 1975) were higher than exports from streams on igneous rock on the Precambrian Shield (4.8 kg km^{-2} yr^{-1}, Dillon and Kirchner 1975) or in the White Mountains (0.95 kg km^{-2} yr^{-1}, Hobbie and Likens 1973). Hence, high P concentrations are more likely to occur in surface and subsurface waters located in regions where soils have developed on sedimentary rock.

Phosphate concentrations in soil solution are primarily determined by 1) mineralization of organic P and 2) weathering of rocks and soils (Tiessen et al. 1984; Illmer and Schinner 1992). In a hardwood forest in New Hampshire, mineralization of organic P added 10 kg ha^{-1} yr^{-1} and weathering added 1.5 to 1.8 kg ha^{-1} yr^{-1} to the labile pool (Bormann and Voigt 1984). Evapotranspiration may also affect P concentrations by decreasing soil water content. Thus, high P concentrations in soil solution may occur during hot, dry periods when evapotranspiration is high. Concentrations in soil solution range from 1 to 270 μg P L^{-1} in mineral soils (clays to sands) and are in equilibrium with soil particles (Hook 1982). P concentrations in soil solution are usually higher in organic and litter layers than in mineral soil layers (e.g. Timmins et al. 1977; Stevens et al. 1989; Yanai 1991). The difference in P concentrations in soil solution between organic and mineral soil layers is likely due to the higher annual contribution from mineralization of organic P than from weathering of soils.

The mobility of P is primarily controlled by the ability of a soil to adsorb or precipitate phosphates. Adsorption occurs rapidly, within hours, while precipitation occurs more slowly (Enfield and Ellis 1982; Sharpley 1983; McBride 1994). Phosphates are usually most soluble in slightly acidic to neutral soils (pH 5.5 to 7) because phosphates precipitate with aluminum and iron sesquioxides in acid soils (pH 3 to 5.5) and with calcium in alkaline soils (pH 7 to 8.5) to form relatively insoluble compounds (Enfield and Ellis 1982; Cresser et al. 1993). However, in the presence of highly charged clays (e.g. smectite, vermiculite) or humus in acid soils, aluminum will rapidly adsorb to clays and organic compounds, and may not react with phosphates (McBride 1994). In this case, phosphate solubility will be high in an acid soil. In the presence of high

amounts of exchangeable sodium and soluble bicarbonates and carbonates in alkaline soils, sodium suppresses calcium phosphate precipitation because bicarbonates and carbonates displace phosphate adsorbed to mineral surfaces (McBride 1994). Phosphate solubility is higher under these conditions than in neutral soils. Phosphates also adsorb to clay particles, limiting P mobility (McBride 1994; Barber 1995). Several studies (e.g. Sharpley 1983; Roberts et al. 1985) found that P content in soil increased with clay content, suggesting that clay adsorbs P. Frossard et al. (1989) also observed that the mobility of phosphates was low in the B horizons of Gray Luvisols where clay and sesquioxide contents were high, but mobility was high in the eluviated A horizon. Hence, phosphates are relatively immobile in soils with high sesquioxide, calcium, or clay content.

Phosphates are often more mobile in organic-rich soils because organic matter has a low adsorption capacity for phosphates (Huang and Schoenau 1997) and competes with P for adsorption sites on mineral colloids (Nagarajah et al. 1970; Singh and Jones 1976). Phosphates are also highly mobile in decaying plant tissue (Qualls et al. 1991, Polglase et al. 1992). However, phosphates may bond to humus in the presence of aluminum or iron (McBride 1994). In a study of boreal soils, P supply rates for plants were greater in the organic horizon than in the eluviated A mineral horizon immediately below it, suggesting that P is more mobile in the organic horizon (Huang and Schoenau 1997). Therefore, the organic layer is likely to contribute more P to subsurface flow than mineral soil layers, but may be lessened by plant uptake of P.

Phosphorus and vegetation

The dominant vegetation can affect P content in the organic layer. In boreal forests of Saskatchewan, P content of forest floors was highest under aspen, lower under mixedwood (aspen and white spruce), and lowest under conifer (black spruce and jack pine) because productivity and thus leaf litter was greatest under aspen (Xiao et al. 1991). P input to soils from litterfall was 3.7 and 6.6 kg P ha^{-1} yr^{-1} in a two-year study of aspen forests in Saskatchewan (Huang and Schoenau 1997). P input from litterfall in other hardwood forests was within a similar range: 3.1 to 8.0 kg P ha^{-1} yr^{-1} in northern U.S. (Likens et al. 1977; Perala and Alban 1982; Zhang and Mitchell 1995). Since deciduous leaves decompose more readily than coniferous needles, there is a higher potential for P to accumulate and be flushed from soils in deciduous forests to adjacent lakes and streams. For example, mean flow-weighted total dissolved P (TDP) concentration of two streams in aspen-dominated forests on the Boreal Plain was high (72 μg L^{-1}, Cooke and Prepas 1998) compared to those from streams in conifer-dominated forests on the Boreal Plain (12 μg L^{-1}, Munn and Prepas 1986), on the Precambrian Shield (9 μg L^{-1}, Nicolson 1988), and in the Cascade mountains (17 μg L^{-1}, Harr and Fredriksen 1988). Therefore, TDP concentrations in subsurface flow are likely to be higher from aspen-dominated than conifer-dominated forests.

Phosphorus and hydrology

Water does not flow evenly through the soil profile because of preferential flow paths, such as cracks, root channels and animal burrows (Mosley 1979; Kachanoski and de Jong 1982; Hill 1996). Water can flow rapidly along preferential flow paths to deeper

soil layers (Mosley 1979; Mulholland et al. 1990; Hill 1996). However, Mulholland et al. (1990) found that when the antecedent soil moisture was high, most preferential flow was lateral along shallow paths (<2.5 m deep) after major rain events. Preferential flow can also occur through a soil layer that is more porous than adjacent soil layers. For example, water flowed laterally through a sandy layer on the surface of clayey B horizon because the B horizon was less permeable than the sand layer (Kachanoski and de Jong 1982). In another study, lateral flow occurred at the interface of the organic layer (humus and litter) and mineral soil because the organic layer was more porous (Pierce 1967). These flow paths may allow subsurface water to flow rapidly into a lake or stream, carrying P with it.

Surface and subsurface runoff can flush out accumulated P in the organic layer and increase P concentrations in runoff (Ng et al. 1993). Models have been developed where accumulated dissolved organic carbon (Boyer et al. 1996) and nitrate (Creed et al. 1996) were flushed from soils when the soil was saturated with water. These nutrients were probably flushed from the organic layer and transported along preferential flow paths. Water flow was probably too rapid along these paths for the nutrients to be immobilized by microorganisms, plants, or soil particles. Thus, nutrients could enter lakes or streams unhindered. If water flowing through a P-rich, organic layer is connected to one of these pathways, a high source of P could be directly linked to a lake or stream. Conversely, deeper groundwater with low P concentrations could be forced to the surface through a similar pathway. For example, if a thick clay layer overlies a sand layer, water would flow up through cracks or root channels in the clay layer. Therefore, water flow paths through the soil are important in the amount of P transported to lakes or streams.

Flooding may also increase P concentrations in shallow subsurface water by introducing anoxic conditions to soils. Anoxia can increase phosphate solubility by reducing iron oxides (D'Angelo and Reddy 1994). Turner and Gilliam (1976) observed that an increase in available P in alkaline soils was due to an increase in soil water content rather than reduction of ferric phosphate because soluble P was controlled by calcium content. Krom and Berner (1980) found that P adsorption coefficients for oxic soils were generally higher (range: 10 to 5000 mL g^{-1}) than anoxic soils (range: 1 to 80 mL g^{-1}), suggesting that oxic soils have a stronger adsorption capacity. In a study of organic soils, P release was higher in flooded than in drained soils (Reddy and Lao 1983). Thus, flooding organic-rich soil layers may elevate P concentrations. Increased P concentrations in soil solution were probably due to a combination of reduction of ferric phosphate, increased soil water content, and flooding of organic-rich soil layers.

Phosphorus and topography

Topography influences water flow and, in turn, influences P distribution and transport. Fine particles and solutes are washed downslope to topographic depressions (Smeck 1985). Depressions often have deeper soils and higher concentrations of solutes, such as phosphates, than hillslopes (King et al. 1983; Smeck 1985; Steila and Pond 1989). Therefore, concave slopes have a greater potential for accumulation of water and nutrients than convex slopes and thus a higher potential for anoxic conditions, which can increase P solubility. For 32 Precambrian Shield lakes, TP and TDP concentrations decreased with greater catchment slope, suggesting that saturated areas and overland flow on saturated soils are important in P transport from terrestrial to aquatic ecosystems

(D'Arcy and Carignan 1997). Therefore, less steep slopes will have more saturated soils, more P for leaching, a longer leaching time, and a greater depth of interaction between runoff and soil.

Phosphorus and climate

Climate determines many aspects of rock and soil weathering and decomposition by controlling precipitation and temperature. Decomposition rates are higher in warm, humid climates because high soil moisture and temperature enhance microbial activity in oxic soils (Wildung et al. 1975; Steila and Pond 1989). Weathering rates are also higher because of the higher soil water content. In addition, temporal variations in local weather can affect P dynamics. During a one-in-one-hundred-year storm event in central Alberta, TP concentrations in the euphotic zone of four deep lakes on the Boreal Plain increased up to 75%, from pre-storm concentrations (Prepas and Trimbee 1988). Rain events may flush P accumulated in low-lying depressions and wetlands during long relatively dry periods (Devito and Dillon 1993). The amount of P flushed from the soil will likely depend on the duration of the dry period and the duration and intensity of the rain event. Around 50% of mean annual precipitation on the Boreal Plain falls in June, July, and August. Thus, flushing of P from soils is likely to occur during the summer. In fact, the majority of annual TP export from first to fourth-order streams was from June to August in two studies on the Boreal Plain (74%, Cooke and Prepas 1998; >85%, Munn and Prepas 1986). In contrast, Singh and Kalra (1975) noted that TP exports from streams in the Foothills of Alberta peaked during snowmelt runoff, which dominated the flow

regime. Therefore, stream P concentrations peaked during high runoff, which can vary in timing, depending on the climate regime.

Forest harvesting effects on hydrology

Forest harvesting often increases water yield primarily due to a decrease in interception and evapotranspiration (reviews in Bosch and Hewlett 1982 and Keenan and Kimmins 1993). These decreases can increase soil moisture, surface and subsurface runoff, and raise the water table in low-lying areas (Williams and Lipscomb 1977, Verry 1986; Keenan and Kimmins 1993). Infiltration can also decrease if heavy machinery compacts the soil and may increase water flow in the organic layer and overland flow. In a review of 94 studies from around the world, Bosch and Hewlett (1982) found that mean water yield increased approximately 40 mm in conifer and eucalypt forests and 25 mm in deciduous hardwood forest with a 10% reduction in forest cover. However, in studies where less than 20% of the watershed was cut, the influence of forest harvesting was difficult to separate from climatic variability. Some studies in coastal regions found no change in water yield (e.g. Cheng et al. 1975; Gilmore et al. 1982) presumably because decreased fog drip counter-balanced any increases in water yield due to decreased evapotranspiration after forest harvesting. Another study observed no change in water yield during drought or incessant rain when soil moisture was similar in both cut and uncut catchments (Leitch and Flinn 1986). Thus, a change in water yield after forest harvesting likely depends on the timing and balance of evapotranspiration and precipitation. Water yield will likely increase when precipitation and evapotranspiration are moderately high and occur at the same time (i.e. peak precipitation occurs in the

summer). If water yield increases after forest harvesting, water flow through the organic layer could potentially increase and flush out P, especially in low-lying areas.

Forest harvesting effects on the phosphorus cycle

Removal of trees could also increase P accumulation because increased soil moisture and temperature could accelerate decomposition in oxic soils and weathering of soils (Fredriksen et al. 1975). Slash and harvest residues also provide more organic matter for decomposition. In several studies, P concentrations in soil solutions did not change in mineral soil layers, yet increased in the organic soil layer after harvesting in deciduous (e.g. Yanai 1991) and conifer forests (e.g. McColl 1978; Adamson and Hornung 1990). Yanai (1991) found that in the organic horizon, mean P concentrations in soil solution were three times higher in a cut than in an uncut watershed in a hardwood forest in New Hampshire. Since P concentrations in soil solution increased only in the organic layer, water flowing through the organic layer will likely increase P input to lakes and streams.

Most research of forest harvesting effects on P dynamics focused on streams and obtained variable results (Table 1.1). Some studies found no detectable change in stream P concentrations or exports due to logging (e.g. Singh and Kalra 1975; Feller and Kimmins 1984; McClurkin et al. 1985; Martin and Harr 1989), while others found increases (e.g. Fredriksen 1971; Nicolson et al. 1982; Blackburn and Wood 1990; Ahtiainen 1992). Increased P concentrations in streams generally occurred only in the first year after harvesting or after site preparation (e.g. scarification, fertilization, slash burning). In contrast to P concentrations, P exports were often higher in cut than in uncut

watersheds for several years after harvesting (e.g. Blackburn and Wood 1990; Ahtiainen 1992). For example, Blackburn and Wood (1990) found mean TP export was almost 6 times (2.4 kg km^{-2}) higher from three cut, sheared, and burned watersheds than from three control watersheds, although stream TP concentrations were not detectably different. TP export was higher due to increased water flow, while P concentrations remained the same due to dilution. The variability in P response to harvesting is likely due to differences in physiographic characteristics and harvest management practices (e.g. percent area cut, timing of cut, site preparation, and buffer strips) (Table 1.2).

Phosphorus and harvest management practices

Differences in percent area cut and timing of cut can vary considerably among studies and regions (Table 1.1). In the literature, the percent area of watershed cut varied from less than 10% to 100%. Therefore, the area contributing to cutting effects can be vastly different. In addition, some forest harvests, like the one in this study, occur when the ground is frozen and soil disturbance is minimal, while others occur when the ground is not frozen and soil disturbance is more likely. When surface soils are disturbed or compacted, overland flow, sediment transport, and leaching of organic matter and nutrients often increase.

Site preparation, such as scarification and herbicide application, commonly used to enhance growth of commercial trees, can potentially increase P losses. Scarification ruptures the soil surface and may accelerate leaching of nutrients. In Finland, scarification caused 3 to 4-fold increase in TP concentrations in a stream without a buffer strip during spring runoff (Ahtiainen 1992). Herbicides suppress growth of brush species, which compete for light, water, and nutrients in the early stages of conifer

growth (Fredriksen et al. 1975). Reduced plant growth decreases P uptake and can potentially increase P leached from the soil. Both scarification and herbicides are used in the Boreal Mixedwood Ecoregion, primarily in conifer-dominated stands.

Vegetated buffer strips between cut blocks and surface waters can potentially reduce P input from cut blocks to adjacent lakes or streams although few studies have focused on buffers. In the first year after clearcutting, a study in Finland observed no detectable change in TDP export from a stream with a 30-m buffer strip, yet observed a 4-fold increase in TDP export from a stream without a buffer (Ahtiainen 1992). Thus, plant uptake of P in the buffer may have reduced P export from the cutblock to the stream. Even after scarification, TDP exports from the stream with the 30-m buffer strip were not detectably different. This study was done in shallow peatlands where the water table was likely close to the surface and flowed through the rooting zone. In deeper soils, water may flow beneath the rooting zone, bypassing buffer vegetation (Hill 1996). Thus, water flow through buffer strips may influence its efficiency in removing P in subsurface flow.

Although buffers are commonly used in both forestry and agriculture, there is much debate about effective buffer strip width. Some buffer strips have trapped up to 99% of TP from agricultural runoff (Castelle et al. 1994). Most of the trapped P is particulate P attached to soil particles. In forestry, transport of soil particles mainly occurs along logging roads or where site preparation disturbs or exposes the soil (e.g. scarification). Therefore, if care is taken to maximize the distance between logging roads (or areas where there is soil disturbance) and lakes, and streams, transport of P from cut blocks by sedimentation will be minimal. However, the transport of soluble P and the

efficacy of buffer strips in reducing soluble P are not well known. In the selected studies on logging effects on P in streams, buffer widths varied from 6 to 30 m (Table 1.1). Some of these studies showed no change in stream P concentrations, while others showed an increase. Differences in the response of stream P concentrations may be due to differences in the ability of the buffer to remove P from subsurface flow because of different buffer widths and water flow paths through the buffer.

Potential forest harvesting effects on phosphorus in lakes

Forest harvesting can disrupt P and hydrologic cycles by increasing weathering, decomposition, and water yield, and thus, can potentially increase P concentrations in adjacent lakes. Phosphorus concentration often limits algal and macrophyte growth in temperate lakes (Schindler 1977; Sharpley et al. 1994). Therefore, elevated P inputs to lakes can change water quality, compromise drinking water supplies, and affect wildlife, fisheries, and recreational activities (Sharpley et al. 1994). Elevated bioavailable P inputs can alter algal communities, altering food webs and influencing higher trophic levels (Sharpley et al. 1994). Many lakes on the Boreal Plain have a relatively high P to nitrogen (N) ratio (Mitchell and Prepas 1990); thus, a small increase in biologically available P with little increase in N may shift algal species to cyanobacteria, which can fix N (Prepas and Trimbee 1988). Cyanobacteria can produce hepato- and neurotoxins that are harmful to wildlife and livestock (Kotak et al. 1993). Thus, additional P input from subsurface flow can potentially change the aquatic ecosystem.

Scope of project

The boreal forest makes up 82% of Canada's forests and is primarily conifer-dominated; however, more than 200 000 km^2 is classified as boreal mixedwood (aspen-balsam poplar-white spruce) (Strong and Leggat 1992). Most of the boreal mixedwood forest is on the Boreal Plain. The soils are P-rich with slightly acidic to slightly alkaline pH (Strong and LaRoi 1985; Huang and Schoenau 1996), and thus, P is likely more mobile than in more acidic or alkaline soils. Deep loamy tills (>30 m deep) underlie most of the region, although tills are shallow (3-5 m deep) near the Canadian Shield and Rockies. Most studies on the effects of logging were conducted in areas with P-poor soils (e.g. Yanai 1991) and shallow overburden or tills (<5 m deep). Annual precipitation in the boreal mixedwood forest is low (220-600 mm) compared to most other forested regions (>750 mm). In addition, approximately 50% of the annual precipitation is from June to August, and long relatively dry periods often precede summer rainstorms. P may accumulate during drier periods and be flushed from soils during subsequent rainstorms. Forest floor masses are also higher in the boreal forest than in temperate and tropical forests (Huang and Schoenau 1996). Hence, there is a high potential for P concentrations in, and export from, subsurface flow to increase after forest harvesting due to flushing of P from the forest floor (or organic layer), especially in low-lying areas.

Although there is a high potential for forest harvesting to increase P in subsurface water in the boreal mixedwood forest, few data are available on the effects of logging on P dynamics in this region. On the Boreal Plain, a small increase in biologically available P may shift algae species to cyanobacteria, which can be toxic to wildlife (Prepas and Trimbee 1988; Kotak et al. 1993). Thus, forest harvesting could alter water quality in the

boreal mixedwood forest. However, the varying responses of P in streams to forest harvesting reported may be due to differences in physiographic characteristics and harvest management practices. Since the boreal mixedwood forest has a unique combination of physiographic characteristics and a high potential for flushing P from soils to lakes or streams, harvesting effects, including efficacy of buffer strips, on P in subsurface water need to be studied.

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Table 1.1: Harvest management practices of selected studies and difference in phosphorus concentration and export in streams due to forest harvesting

Location [a]	Catchment Area (ha)	Cut area (%)	Harvest management practices: cut type [b]	time of cut	buffer (m)	Other [c]	Export (kg ha^{-1} yr^{-1}): TP [d]	TDP [d]	Concentration (µg L^{-1}): TP [d]	TDP [d]
Canada										
3 West central Alberta	20-100						-	n.s	-	n.s.
4 Haney, B.C.	23-68	19-61	cc	summer-fall	0	br	n.s.	n.s.	n.s.	n.s.
5 Lake Nipigon, ON	102-438	50-100		fall-winter	0-30		-	-	+3 [1]	+3 [1]
6 Experimental Lakes Area, ON	35-170	100		?	?	sc	+0.08	-	-	+25 [1]
United States										
7 Alto, Texas	2.6-2.7		cc, sh	summer	6-20	br	+0.32 [1]	+0.036 [1]	+142 [1]	+37 [1]
			cc, ch	summer	6-20	br	n.s.	n.s.	n.s.	n.s.
8 Lexington, TN	0.17-0.56		cc		yes		n.s.	-	n.s.	-
9 Clemson Experimental Forest, SC	0.48-2.18					br				
10 Hubbard Brook, NH	13-16		cc	fall-winter	?	h	+0.02	-	-	-
11 Fernow Experimental Forest, WV	34-38		cc	all	10-20	h	0.25	-	-	-
12 Mt. Hood National Forest, OR		15-17	cc	summer	sparse	br,sc				
13 H.J. Andrews Experimental Forest, OR	13-21.4	100	cc, sc	summer	?	br	-	n.s	-	n.s.
14 H.J. Andrews Experimental Forest, OR		25-100	pc,cc	?	?	before br	-	+0.09	-	n.s.
						br	-	+0.40 [2]	-	+23 [2]
15 Blue Mountains, OR	30-118	17-100	pc,sc	late summer	?	br,sc	-	-	-	n.s.
	24	41	cc	late summer		br,sc	-	-	-	+8
16 Coast Range, OR		25-100		spring-summer		br	-	-	n.s.	-
17 Silver Creek, ID	110-163	23	cc	fall	15-30	br	n.s.	-	-	-
Europe										
18 Nurmes, Finland	494	58	cc	winter	0	sc	+0.51 [3]	+0.36 [3]	+130 [3]	-
	0.54	56	cc	winter	30	sc	+0.02 [3]	n.s.	-	-

[a] 3) Singh and Kalra 1975, 4) Feller and Kimmins 1984; 5) Nicolson 1988; 6) Nicolson et al. 1982; 7) Blackburn and Wood 1990; 8) McClurkin et al. 1985; 9) Van Lear et al. 1985; 10) Hobbie and Likens 1973; 11) Aubertin and Patric 1974; 12) Harr and Fredriksen 1988; 13) Martin and Harr 1989; 14) Fredriksen 1971; 15) Tiedemann et al. 1988; 16) Brown et al. 1973; 17) Clayton and Kennedy 1985; 18) Ahtialnen 1992.

[b] cc, clearcut; ch, chopping; pc, patch cut; sc, select cut; sh, shearing

[c] br, slash burning; h, herbicides; sc, scarification

[d] n.s., not significant; changes in export and concentration recorded are statistically significant ($P<0.05$)

[1] first post-harvest year only; [2] first year after slash buring; [3] mean of three years post-harvest before scarification

Table 1.2: Potential for phosphorus export from cutblocks depending on physiographic characteristics and harvest management practices

		potential for P export	Comment
Physiographic characteristic			
geology		sedimentary > igneous	
		glacial till > granite	
soils	organic content	organic-rich > organic-poor	
	texture	sand > clay	if water flowing into the layer is high in P
		clay > sand	if water flowing into the layer is low in P
	pH	neutral / slightly acidic > acidic / alkaline	
	salinity	saline > non-saline	
vegetation		aspen > white spruce > jack pine / black spruce	
hydrology	water table	high > low	
	preferential flow paths	lots > few	
	flooding	yes > no	
topography	slope	low > high	
		concave > convex	
climate	evapotranspiration (ET)	high > low	
	precipitation	high > low	
	timing of ET and precipitation	high at same time > very wet / very dry	
Harvest management practices			
% area cut		greater with increase % cut	> 20% of watershed cut
timing of cut		summer > winter	if ground freezes in winter
		wet season > dry season	
soil disturbance		yes > no	
fertilization (with P)		yes > no	
proximity of roads		closer > further away	
buffers		?	

Table 1.3: Climate and watershed characteristics in uncut watersheds of selected studies on logging effects on phosphorus compared to the Boreal Mixedwood Ecoregion of Alberta

Location [a]		Climate [b]	Precipitation annual (mm)	Precipitation distribution [c]	Bedrock [d]	Overburden and depth (m)	Soil texture [e]	Slope (%)	Dominant Vegetation [f]
Boreal Mixedwood Ecoregion									
1	Baptiste Lake, AB	cs, cw	500	50% summer	S	till, 30-100	CL,L,SL	4-7	Po, Pic
2	Two Creek and Sakwatamau, AB	cs, cw	570	50% summer	S	till, 3-5	C,CL		Pic, Pin, Po
19	northern Alberta	cs, cw	220-600	50% summer	S	till, 100-200	C,CL-SL	4-40	Po, Pic
3	west central Alberta (incl. foothills)	cs, cw	390-570	spring, summer		till			Pin, Pic, Po
Canada									
4	Haney, B.C.	cs, mw	2200-2700	even, rain	I	till, shallow	SL,LS		Ts, Th, Ps
5	near Lake Nipigon, ON	ws, cw	750	40% summer	I, M	till, shallow	SL,L	low	Pic
6	Experimental Lakes Area, ON	ws, cw	500-750	43% summer	I	shallow		low	Pin, Pic
United States									
7	Alto, TX	ws, mw	1070	even	S		C,SL	4-25	Pin, Hd
8	Lexington, TN					loess	SL		
9	Clemson Experimental Forest, SC		960-1400		I		L,C	10-16	Pin
10	Hubbard Brook, NH	cs, cw	1230	even	I,M	till, 0.2	SL	21->70	Be, Ac
11	Fernow Experimental Forest, WV		1470	even	S		L	10-65	Ac, Po, Pr
12	Mt. Hood National Forest, OR	ws, mw	2620	winter	I		L	5-50	Ab, Ts, Ps
13, 14	H.J. Andrews Experimental Forest, OR	cs, mw	2190	even, rain	I	ash	SL,L,CL	20-40	Ps
15	Blue Mountains, OR	cs, cw	730-1900	87% snow	I	ash		2-25	Ab, Pic, L, Ps, Pin
16	Coast Range, OR	cs, mw	2540	winter, rain	S		L	35-50	Al, Ps
17	Silver Creek, ID	cs, cw	650-800	winter		<1 m	S	steep	Ps,Pin
Europe									
18	Nurmes, Finland		540	50% snow	I			4 to 6	

[a] 1) Cooke and Prepas 1998; 2) Munn and Prepas 1986; 3) Singh and Kalra 1975, 4) Feller and Kimmins 1984; 5) Nicolson 1988; 6) Nicolson et al. 1982; 7) Blackburn and Wood 1990; 8) McClurkin et al. 1985; 9) Van Lear et al. 1985; 10) Hobbie and Likens 1973; 11) Aubertin and Patric 1974; 12) Harr and Fredriksen 1988; 13) Martin and Harr 1989; 14) Fredriksen 1971; 15) Tiedemann et al. 1988; 16) Brown et al. 1973; 17) Clayton and Kennedy 1985; 18) Ahtiainen 1992; 19) Strong and Leggat 1992.

[b] cw, cool summers; ws, warm summers; mw, mild winters; cw, cold winters

[c] summer = June to August; winter = December to February

[d] I, igneous; S, sedimentary; M, metamorphic

[e] C, clay; CL, clay loam; SL, sandy loam; L, loam; St, stony

[f] Hd, hardwood species; Ab, *Abies*; Ac, *Acer*; Al, *Alnus*; Be, Betula; Pic, *Picea*; Pin, *Pinus*; Po, *Populus*; Pr, *Prunus*; Ps, *Pseudotsuga*; L, *Larix*; Th, *Thuga*; Ts, *Tsuga*

Chapter 2 Dissolved phosphorus in shallow subsurface waters in an uncut and cut subcatchment of a lake on the Boreal Plain[1]

Introduction

Forest harvesting can affect the phosphorus (P) cycle because of potential increases in water yield, decomposition, and weathering of soils as well as decreases in plant uptake of P. In a review of 94 studies, Bosch and Hewlett (1982) found that mean water yield in pine, eucalypt, and deciduous forests increased with decreased forest cover, where more than 20% of the watershed was cut. Increased water yield was due to decreased interception and evapotranspiration, enhancing weathering and transport of P from soils (Bosch and Hewlett 1982; Anderson 1988). Furthermore, P mineralization and accumulation may increase due to increased soil moisture and temperature, the addition of slash residue, and decrease in plant uptake of P (Fredriksen et al. 1975; Nicolson et al. 1982). Some studies report increased P concentrations in, and exports from, streams (e.g. Nicolson et al. 1982; Blackburn and Wood 1990; Ahtiainen 1992), while others report no detectable change (e.g. Feller and Kimmins 1984; McClurkin et al. 1985; Martin and Harr 1989). The different results may be due to differences in harvest management practices (e.g. buffer strips) and in physiographic characteristics (e.g. soils, geology, climate, topography, vegetation). Several studies of soil solutions found increased P concentrations in the organic layer or forest floor, yet found no detectable change in mineral soil layers after logging (e.g. McColl 1978; Stevens and Hornung 1988; Yanai 1991). Hence, more P may accumulate and be flushed from soils, especially the organic layer, after logging, increasing P export to adjacent lakes and changing the water quality.

[1] A version of this chapter will be submitted for publication.

Forest harvesting on the Boreal Plain has increased in the past decade, yet few data are available on the effects of logging on P dynamics in this region. The forests are predominantly mixedwood and cover over 200 000 km^2 (Strong and Leggat 1992). The Boreal Plain generally has P-rich soils with deep loamy tills (>30 m), whereas most studies on the effects of logging have been conducted in regions with P-poor soils and shallow overburden or till layers (<3 m). On the Boreal Plain, around 50% of the annual precipitation falls in the summer and is often preceded by relatively dry months (Atmospheric Environment Service 1998). Hence, P may accumulate during the dry months and be flushed from soils during summer rainstorms when runoff is high, especially from low-lying areas, such as wetlands and riparian zones (Devito and Dillon 1993; Creed et al. 1996). Thus, there is a high potential for flushing of P from cutblocks to lakes. However, buffer strips may reduce P transport from cutblocks by plant uptake of soluble P or by trapping sediments with high particulate P content (Ahtiainen 1992; Castelle et al. 1994). Sediment transport, which increases with soil exposure and disturbance, was minimal at the study sites (E. Prepas, unpublished data). For buffers to reduce soluble P in subsurface flow, water has to flow through the rooting zone where plants take up P. However, preferential flow paths (e.g. root channels, animal burrows, sand lenses) may allow water to flow deeper into the soil (Mosley 1979; Hill 1996). Therefore, studies are needed to evaluate the effects of logging, including buffer strips, on P in subsurface water on the Boreal Plain because of its unique combination of physiographic characteristics, large area, and high potential for flushing of P from soils in low-lying areas to lakes and streams.

The objectives of this study were to assess 1) spatial and temporal trends in total dissolved phosphorus (TDP) in shallow subsurface water (< 2.5 m deep) and 2) the effect of forest harvesting and a vegetated buffer on these trends. Shallow subsurface water was sampled because 1) the primary source of P likely to be affected by forest harvesting is the organic layer and 2) clay and calcium content, which react with and immobilize P, usually increase with depth in soils (Frossard et al. 1989). Water was collected from transects of nested piezometers and wells within 100 m of a boreal lake in an uncut and cut subcatchment. The hypotheses tested in this study were that TDP concentrations in shallow subsurface water would 1) decrease with depth, 2) increase when water flowed through the organic layer, 3) be higher in the cutblock than in the uncut subcatchment, and 4) decrease in a buffer strip with a mean width of 23 m.

Methods

Study Site

The study was part of the Terrestrial and Riparian Organisms, Lakes and Streams (TROLS) project, an interdisciplinary research project based at the University of Alberta. TROLS studies how riparian buffer strips of differing widths (20, 100, 200, and 800 m) affect various aspects of terrestrial and aquatic ecosystems in the mixedwood forest on the Boreal Plain. Current Operating Ground Rules require a 30 to 100-m wide buffer around permanent lakes. The twelve TROLS lakes are in 3 locations with 4 lakes; each lake has a different buffer width. The lakes are located in Alberta-Pacific Forest Industries, Inc. and Weyerhauser Canada, Ltd. Forest Management Areas and are near Smith, Slave Lake, and Lac La Biche, Alberta.

This study was conducted in two subcatchments (SCs) of Moose Lake (55°09'N, 111°45'W), the 20-m buffer treatment lake in the Lac La Biche region (Fig. 2.1). Drainage basin area is approximately 575 ha with a mean slope of 8%. Topography is level to rolling, typical of the Boreal Plain. Mean drainage basin slope was higher than the mean for the 11 other TROLS lakes (mean: 4%; range: 1 to 5%). Soils are predominantly Gray Luvisols and Eutric Brunisols in uplands and Gleysols in lowlands. Primary surficial deposits are loamy glacial till, 125 to 175 m thick, underlain by shale and ironstone of the La Biche Formation (Alberta Geological Survey 1972). Climate is continental with warm summers (mean July temperature: 18°C) and cold winters (mean January temperature: -17°C) (Atmospheric Environment Service 1998). Long-term (1971-1997) mean annual precipitation and evapotranspiration (ET) are 468 and 514 mm, respectively. Precipitation and ET values are weighted means of the values for Atmospheric Environment Service stations in Athabasca, Slave Lake, Fort McMurray, and Cold Lake. ETs were estimated from mean monthly temperatures and day length by Thornthwaite's method (Hewlett 1982). Snow accounts for 25 to 30% of annual precipitation. Annual precipitation was 576 and 487 mm and annual ET was 452 and 482 mm for 1996 and 1997, respectively. Precipitation was higher than average during the summer (June to September) of 1996 and in June and September 1997, but was close to the mean for the rest of 1996 and 1997. Annual discharge of the Owl River, a local stream, was less than 90 $m^3 yr^{-1}$ in each of the previous 13 years to the study, yet discharge was 185 and 239 $m^3 yr^{-1}$ during the two study years (1996 and 1997, respectively) (Water Survey of Canada 1998). Therefore, the study was mainly

conducted during a period of high runoff when annual precipitation was greater than estimated ET.

The uncut and cut SCs were chosen on the basis of similar aspect, vegetation, and mean slope prior to logging of the cut SC in 1996 (Table 2.1; Fig. 2.1). The SCs were identified by their topographic boundaries although some subsurface water may be contributed from outside these boundaries. Mean slopes of the draws of the SCs (8 and 11 %) were similar to the mean slope of the lake drainage basin (8%). Percent conifer cover was higher along the center transect in the SCs (33 to 39%) than in the lake basin (7%), as ephemeral draws tend to have more conifer than uplands on the Boreal Plain. Although percent conifer cover in the draws was relatively high, it was within the range of percent watershed cover in the other TROLS lake basins (1 to 42%). Vegetation was typical of the boreal mixedwood forest, comprising of aspen (*Populus tremuloides*), balsam poplar (*P. balsamifera*), white spruce (*Picea mariana*), and some paper birch (*Betula papyrifera*). Willow (*Salix* sp.) and alder (*Alnus* sp.) grew along the shore of the uncut SC. Prior to forest harvesting, an alder stand grew at site F9 in the cut SC. Forest harvesting occurred in the winter of 1996-97, and a treed buffer strip with a mean width of 23 m was left along the shore of the cut SC.

Instrumentation and characterization of subcatchments

Three transects of wells and piezometers were installed in each SC (Fig. 2.2 and 2.3). The main transect was in the center of the topographic low of both SCs. Most sites along the transects consisted of nested piezometers and a well (Fig 2.4). Some wells and piezometers were installed in the uncut and cut SC in 1996; the rest were installed in 1997. Wells consisted of 2-m long polyvinylchloride (PVC) pipe (Schedule 40, inner

diameter: 51 mm) with slots (3 mm wide, 24 mm long) every 7.5 cm on either side. Piezometers consisted of a 17.5-cm long screened (slots: 0.010 mm, 6 mm apart) PVC (Schedule 40, inner diameter: 19 mm) piece wrapped in spunbond polyester fabric (NILEX MD7407; opening size <100 μm) attached to solid PVC pipe with a coupling. Wells were nested with piezometers, except for the well closest to the lake in each transect. Wells were 1.1 to 1.9 m deep. Most piezometers were between 0.9 to 2.5 m deep. Three nests were installed approximately 4 m apart from the first well in all transects. In the center transect, four additional nests were installed: 3 nests were approximately 4 m apart between 20 and 36 m and one nest about 50 m from shore. Single piezometers were installed along the center transect greater than 50 m from shore and near the breaking point of the hillslope. The topography, wells, and piezometers were surveyed with a total station (Sokkia 3B-II) in October 1997.

The general lithology was recorded while installing the wells and piezometers. The lithology was interpolated between sites. The organic layer includes leaf litter, fibric and humic matter. The hand texture method was used to identify general soil textures: loamy sand to sand (S), sandy loams to loams (SL), clay loams to sandy clay (CL), and clay (C) (Steila and Pond 1989).

Pressure heads and groundwater levels were used to create a flow net for each sampling date. Equipotential lines were drawn among the points by triangulation and hand. Groundwater flow was assumed to be perpendicular to these lines with the assumption that the soils are isotropic and homogeneous.

Saturated hydraulic conductivities (K_{sats}) were determined for each piezometer by Hroslev's method (Freeze and Cherry 1979). Some piezometers had extremely slow

recovery rates (>4 days) and are recorded with K_{sats} less than 10^{-9} m d^{-1}. K_{sat} (m d^{-1}) was defined as:

$$K_{sat} = \frac{r^2 \ln(L/r)}{2Lt_o}$$

where r is the radius of PVC pipe, L is the length of screened PVC pipe, and t_o is the basic time lag (time for water to recover 37%). Mean K_{sats} for each mineral soil type were determined from K_{sats} of piezometers.

Transmissivity is the amount of water flowing horizontally through a fully saturated layer of homogeneous soil under a hydraulic gradient of 1 (Fetter 1994). The total transmissivity (m^2 d^{-1}) of the wells was determined for each sampling date with the following equation (Fetter 1994):

$$T = \Sigma(b_{st}K_{st})$$

where b_{st} (m) is the portion of saturated soil type (organic, S, SL, CL, and C) intersected by the well and K_{st} (m d^{-1}) is the mean K_{sat} for the soil type. Mean bulk density (0.13 g cm^{-3}) of 41 organic layer samples from the Moose Lake catchment (N. Carmosini, University of Alberta, unpublished data) was in the low range of bulk densities (0.02 to 1.45 g cm^{-3}) for organic soils reported by Slaughter and Kane (1979). Reported K_{sats} of organic layers range from 0.4 to 121 m d^{-1} (Mosley 1979, Slaughter and Kane 1979, Binley and Beven 1992). Therefore, a K_{sat} of 100 m d^{-1} was used for the organic layer because of the low bulk density of the organic layer at Moose Lake.

Field sampling

Well water was sampled during and after major rain events (>10 mm) during July and August 1996 and biweekly in September 1996. Wells were sampled approximately every two weeks from late March to early October 1997. Piezometers were sampled once in May and approximately every two weeks from July to October 1997. Groundwater levels in wells and pressure heads in piezometers were measured prior to purging wells and piezometers. Groundwater levels were measured with a plastic float attached to a string, marked every 1.0 cm. Pressure heads were measured by blowing through a polyethylene tube, marked every 0.5 cm, until water bubbles were heard. Wells and piezometers were purged the day before sampling due to slow recovery rates in several of the wells and piezometers. Wells were sampled with a bailer attached to a stainless steel chain. Piezometers were sampled with a tube attached to a Nalgene Erlenmeyer flask and hand vacuum pump or to a syringe with a three-way valve. The bailer, flask, and syringe were washed in a 3% hydrochloric acid bath and rinsed with deionized distilled water between sampling dates. Collection devices were rinsed with a small amount (10 to 50 mL) of sample before collection. Water samples were kept on ice in the field and refrigerated at 4°C for 24 to 48 h until prepared for analysis.

Physical and chemical analyses of water samples

Water samples were measured for temperature, electrical conductance, and pH, and analyzed for TDP, dissolved organic carbon (DOC), and cations (Ca, Mg, K, and Na). Cations were done only in 1997. In 1997, samples from wells were also measured for dissolved oxygen (DO) with a DO meter in the field. Specific conductance and

temperature were measured with an electrical conductance meter (ORION Model 130) in the field. The pH was measured with a pH meter (ORION Model 250A) within 12 h of sampling. Water samples were filtered through a pre-rinsed Gelman glass filter (average pore size: 1.2μm) prior to analysis of DOC and cations and through a pre-rinsed 0.45-μm Millipore HAWP membrane filter for TDP. Cation samples were acidified to a pH of 3 with nitric acid. DOC concentrations were determined by combustion with an Ionics 1505 programmable carbon analyzer. Cation concentrations were measured with a Perkin Elmer 3300 Atomic Absorption Spectrophotometer. Magnesium and calcium were determined by atomic absorption. Potassium and sodium were determined by emission. TDP was analyzed according to Menzel and Corwin's (1965) potassium persulfate method as modified by Prepas and Rigler (1982). All water samples were filtered and prepared within 48 h of collection, then stored at 4°C. Mean concentration of duplicates was reported for each sample.

Estimation of TDP export and export coefficient

TDP exports and export coefficients were estimated for the flow at the base of the SCs (F2 and G2) for each sampling date from June to September 1997. TDP exports (mg d^{-1}) were estimated by taking the product of the slope of the water table (dh/dx), mean TDP concentration (C, mg m^{-3}) and transmissivity (T, m^2 d^{-1}) for the wells at either F2 or G2, and distance (d, m) across the draw of the SCs at F2 and G2.

$$\text{Export} = (dh/dx)\ C\ T\ d$$

The distances for the uncut and cut SC were 35 and 22 m, respectively. The assumptions were: 1) transmissivities and concentrations in the wells at F2 and G2 were representative of water flowing from the SCs and 2) water paths were the same across F2 and G2. TDP export coefficients ($\mu g\ m^{-2}\ d^{-1}$) were estimated by dividing the export ($\mu g\ d^{-1}$) by the SC area (m^2). These export coefficients were compared with daily export coefficients for the summer derived from two studies on streams in the boreal mixedwood forest (Munn and Prepas 1986 and Cooke and Prepas 1998).

Statistical Analyses

Wells and piezometers in each SC were treated as separate groups because water samples from piezometers were discrete samples from mineral soils, whereas samples from wells were composite samples from several soil layers. Non-parametric tests were used throughout because the assumptions of normality and homogeneity of variance were not met. The levels of significance for Spearman correlation coefficients (r) were adjusted by Bonferroni's equation to account for multiple parameters tested. Since Ca and Mg were strongly correlated ($r>0.95$, $P<<0.001$), correlations with Ca, but not Mg, were included. Temperature data for piezometer samples were not included in the analyses because the temperature often increased rapidly after sampling due to high air temperature. Temperatures for well samples were more stable because of the larger volume and were therefore included in the analyses.

Result

Lithology

The surficial lithology consisted of interlayers of glacial till and sand in both SCs (Fig. 2.5 and 2.6). These layers were often discontinuous, leading to considerable spatial variability. The organic layer in the uplands of the cut SC (F9: 39 cm) was almost double that of the uncut SC (G9: 20 cm). Near the lake, the organic layer of the cut SC (F1: 25 cm) was less than half that of the uncut SC (G1: 61 cm). Both SCs had a large sandy layer in the uplands and many interlayers of sand and clay till near the shore. A shallow (1 to 2 m deep), broad layer of clay underlay the upper soil layers in the cut SC. In contrast, the uncut SC had less clay in the upper two meters of the soil.

Hydrology

Mean K_{sats} of the mineral soil layers were within the range for K_{sats} of similar soil types in the literature (Table 2.2). Mean K_{sats} were in the low end of the range for sand and in the upper end of the range for clay. Mean K_{sats} differed by at least one order of magnitude between each soil type. Transmissivities of 27 wells in both SCs ranged from 1.3×10^{-7} to 71 $m^2 d^{-1}$ (Table 2.3). Transmissivities in individual wells increased up to 6 orders of magnitude when water flowed through the organic layer. Mean transmissivity of all the wells was two orders of magnitude higher when the water table intersected the organic layer than when it did not.

Although both sites showed a predominance of lateral flow from May through October 1997, the two SCs had different groundwater table profiles (Fig. 2.5 and 2.6). The water table was consistently high at F9 in the cut SC (mean depth: 0.05 ± 0.01 m),

varying only 7 cm from July to September 1997. Although no data are available for 1996, this area supported alders, which prefer saturated soil conditions, prior to logging. Mean water table depth was deeper at F4 (0.36 ± 0.13 m) and became shallower at F1 (mean depth: 0.10 ± 0.03 m). In contrast to the cut SC, the water table in the uplands (G9) of the uncut SC (mean depth: 1.01 ± 0.13 m) was almost 1 m deeper than at F9 and became shallower approaching the lake (G1: mean depth: 0.09 ± 0.05 m). Mean depths were for the period of July 15 to September 23, 1997. Water table fluctuations were dynamic, dropping between 14 and 76 cm at all the sites (excluding F9) within a 4-week period.

Trends along the hillslope

TDP concentrations in most wells were higher than, or similar to, concentrations in adjacent piezometers (Fig. 2.7 and 2.8). Concentrations in piezometers generally decreased with depth. For example, TDP concentrations in the nested piezometers at F3 decreased from 285 to 75 $\mu g\ L^{-1}$ on July 29, 1997, while TDP in the well was 426 $\mu g\ L^{-1}$. One notable exception was at F9, where concentrations in the piezometer at 2.0 m were approximately 100 $\mu g\ L^{-1}$ more than concentrations in the piezometer at 1.3 m throughout the 1997 season. Furthermore, mean TDP concentrations in both SCs were higher in piezometers above than those below a depth of 1.7 m (Mann-Whitney test, $P<0.01$). The lowest TDP concentrations in piezometers were usually in clay.

Total dissolved P concentrations generally increased when the water table rose after summer rainstorms and intersected the organic layer (Fig. 2.7 and 2.8). For example, TDP concentration in the well at F7 was an order of magnitude higher when the

water table intersected the organic layer (Fig. 2.7). However, TDP concentrations were also high in the wells from F2 to F4 when the water table was low although the water table does not intersect the organic layer. Wells from F2 to F4 intersected a sandy layer that appeared to connect to the organic layer (Fig. 2.5). TDP concentrations in wells were also negatively correlated with the depth to the water table and positively correlated with transmissivity, DOC, and K (Table 2.4, Fig. 2.9). There were, however, apparent thresholds around 0.4 m for water table depth and 60 mg C L^{-1} for DOC below which there was no relationship. In the mineral soil layers (piezometers), there was virtually no relationship between TDP and DOC concentrations, which were mostly lower than 60 mg C L^{-1} (r^2<0.16; Fig 2.10b; Table 2.5). There was also no relationship between TDP and K (r^2<0.14, not shown) or TDP and K_{sat} in the piezometers (r^2<0.12; Fig. 2.10f; Table 2.5). Thus, TDP concentrations in wells, which may have water contributed from both mineral soil and organic layers, were related to DOC, K, and well transmissivity. In contrast, TDP concentrations in piezometers, which were only in the mineral soil layers, were not related to DOC, K, or K_{sat}.

The relationship between TDP and the other major cations (Mg, Ca, Na) was stronger in piezometers (r^2>0.5) than in wells (r^2<0.3) (Table 2.4 and 2.5; Fig. 2.9 and 2.10). Mg, Ca, and Na concentrations were negatively correlated with TDP concentrations in wells of the cut SC and in piezometers, but were positively correlated with TDP in wells of the uncut SC. Specific conductance and TDP concentrations were also negatively related in wells of the cut SC and in piezometers. However, specific conductance and TDP in wells were not related in the uncut SC (r^2=0.05; Table 2.4) and

were weakly related in the piezometers (r^2=0.14; Fig 2.10), suggesting that other ions may have contributed to specific conductance in the uncut SC.

Fluctuations in estimated TDP exports coincided with fluctuation in the water table. TDP exports from the hillslope (at F2 and G2) were lowest when the water table was low on August 12, 1997 (Table 2.6). Mean daily TDP export was higher in the uncut than the cut SC; however, mean export coefficients were similar. These coefficients were remarkably similar to studies of streams on the Boreal Plain.

Forest harvesting and TDP

TDP concentrations in wells were higher in the cutblock than in the uplands of the uncut SC (Mann-Whitney test, P<0.05), especially when the water table was high (Fig. 2.7 and 2.8). Shallow piezometers (<1.7 m deep) had similar mean TDP concentrations in the two SCs (Mann-Whitney test, P=0.96). However, mean TDP concentration in deeper piezometers (>1.7 m deep) was about 1.5 times (46 $\mu g\ L^{-1}$) higher in the uncut than in the cut SC, which had more clay content in the soil (Mann-Whitney test, P<0.05). In the buffer strip, TDP concentrations were similar (100-200 $\mu g\ L^{-1}$) from F7 to F2, except at F4, when the water table was low and below the organic layer. However, TDP concentrations in wells decreased by an order of magnitude with distance from the cutblock into the buffer when the water table was high.

Trends near the lake

Mean TDP concentrations of the three wells at the site closest to shore in the cut (F1) and uncut (G1) SC generally increased over the summer and peaked after major rain events (Fig 2.11), similar to most of the wells along the hillslope. From July 18 to

August 22, 1996, TDP concentrations increased by 37 and 28 times at F1 and G1, respectively. Most of the peak occurred after two large rain events (August 4 to 7: 68 mm; August 13: 24 mm) which flooded G1 for 10 days, but did not flood F1. An additional, smaller TDP peak occurred at G1 on September 11, 1996. After consecutive rain storms in June 1997 (June 18 to 19: 45 mm; June 22 to 24: 43 mm), TDP concentrations increased by 3 and 6-fold at F1 and G1, respectively. Mean TDP concentrations at F1 only peaked at 96 $\mu g\ L^{-1}$ on July 15, whereas concentrations at G1 peaked at 976 $\mu g\ L^{-1}$ two weeks later, on July 29. Site G1 was flooded for 39 days, over two times longer than F1, which was flooded for 16 days. An additional peak occurred at both sites, but at different times in the season. On April 22, 1997, TDP concentrations at G1 peaked at 317 $\mu g\ L^{-1}$, during snowmelt when there was standing water around G1. Concentrations at F1 peaked at 108 $\mu g\ L^{-1}$ on September 9, 1997, 3 days after a rainstorm. Thus, TDP concentrations in wells at F1 and G1 peaked at the same time in 1996, yet peaked at different times in 1997.

DOC concentrations in both SCs generally increased over the summer and increased after the major rain event in August 1996 (Fig. 2.11). DOC concentrations increased 2 to 3-fold in 1996, several days before TDP concentrations peaked. After August 11, 1996, DOC was not analyzed due to the loss of samples. In 1997, DOC concentrations at G1 increased 8-fold from April 22, 1997 and peaked at 131 $mg\ L^{-1}$ on July 29, the same time as TDP concentrations. In contrast, DOC at F1 gradually increased 3-fold over the growing season and peaked at 51 $mg\ L^{-1}$ at the same time as the second TDP peak in the cut SC on September 9, 1997. However, a peak in DOC concentrations was likely missed after the rainstorms in June 1997. Trends in DOC

concentrations were similar in the cut and uncut SCs in 1996 and the uncut SC in 1997, but differed in the cut SC in 1997.

Discussion

Influence of mineral soil characteristics

The general decrease in TDP concentrations in piezometers with depth was likely due to a combination of increases in Ca, clay, and perhaps iron (Fe) content in the soil because phosphates react with Ca, Fe, and clay particles to form relatively insoluble compounds (Cresser et al. 1993). In the boreal forest of northern Saskatchewan, precipitation of Ca phosphates increased with depth (Frossard et al. 1989). The negative correlation between TDP and Ca concentrations in piezometers (Table 2.6) suggests that Ca may limit P solubility in mineral soil layers.

Several studies report that clay limits the mobility of P (e.g. Sharpley 1983; Frossard et al. 1989). Mean TDP concentrations were highest in shallow piezometers (<1.7 m deep), which were in soils with less clay. Likewise, mean TDP concentration in deep piezometers (>1.7 m deep) were higher in the uncut than in the cut SC because the uncut SC had less clay content in the upper 2.5 m of soil. Thus, clay probably limited P mobility.

Although Fe was not measured in my study, Fe could be a source of P adsorption because Frossard et al. (1989) found that P mobility was low in the B horizon of Gray Luvisols with high Fe content in Northern Saskatchewan. Furthermore, several studies in the region have documented relatively high Fe in lake sediments (e.g. Shaw and Prepas

1990; Manning et al. 1994), indicating a potential for relatively high Fe concentrations in soils. Thus, Fe could be important in limiting P mobility in aerobic soils in this region.

Influence of organic matter and water flow

The mobility of P may also depend on other factors, such as organic matter content. In a study of 60 soil samples from various soil classes, TDP concentrations increased with an increase in organic carbon-clay content ratio (Sharpley 1983). Although TDP concentrations in the wells generally decreased when more clay contributed water to the well, there were many exceptions, especially in the cut SC (Fig. 2.9). These exceptions were likely due to the contribution of water from the organic layer. The positive correlations among K, DOC, and TDP concentrations suggests that subsurface water flowing through the organic layer contributed more P than the mineral layers because the main source of both DOC and K in subsurface water is the organic layer (Chapman et al. 1993; Boyer et al. 1996). Phosphates are also more mobile in organic-rich soil layers because organic matter has a low affinity for binding phosphates (Huang and Schoenau 1997) and competes with P for adsorption sites on mineral colloids (Singh and Jones 1976). Therefore, water contributed to the wells from the organic layer will likely have higher TDP concentrations and weaken the relationship between TDP and clay, Ca, Mg, and Na concentrations, which are associated with mineral soil layers.

Furthermore, when water flowed through the organic layer, mean transmissivities of wells increased by 1 to 2 orders of magnitude (Table 2.3). Some individual transmissivities increased up to 6 orders of magnitude. In one well (G2, right transect), which intersected only clay and organic layers, the estimated transmissivity increased from 1.3×10^{-6} to 8.0×10^{-1} m^2 d^{-1} when only 1 cm of the organic layer was saturated

because of the much higher K_{sat} of the organic layer. Thus, water flowing through the organic layer may contribute more water to wells than underlying mineral soil layers.

Several studies have developed models in which DOC or nitrate accumulated in the organic layer during low flow periods and was flushed from the soil during high flow (e.g. Boyer et al. 1996; Creed et al. 1996). This flushing concept is useful in explaining the spatial and temporal patterns in TDP concentrations in the boreal mixedwood forest. Although the two SCs had different water table profiles (Fig. 2.5 and 2.6), TDP concentrations in wells increased as the water table approached the ground surface and transmissivity increased (Fig. 2.9). For example, when the water table was 17 cm closer to the surface at F7, TDP concentration in the well was an order of magnitude higher, yet concentrations in adjacent piezometers (in the mineral soil layers) were similar on both sampling days (Fig. 2.6). The increase in the well was likely due to water flowing through the organic layer. However, TDP concentrations in wells from F7 to F2 were still relatively high although the water table was below the organic layer. These concentrations were likely maintained by water flowing through the organic layer in the cutblock, then flowing along preferential flow paths (i.e. the sandy layer, Fig. 2.5) to the wells. Therefore, TDP concentrations in wells were higher during high flow, suggesting that subsurface flow flushed P from the soil, similar to studies on other nutrients.

Several studies observed that most dissolved organic P in soil solution was associated with DOC, suggesting that this was a mechanism for leaching of P from soils (Qualls et al. 1991; Donald et al. 1993). Concentrations of DOC and TDP in wells were correlated in both SCs in 1997 and in the cut SC in 1996 (Table 2.4), which supports the idea of DOC possibly mitigating P transport from hillslopes. However, mean DOC

concentrations peaked before mean TDP concentrations for daily samples of the wells at F1 and G1 during and after consecutive storm events in August 1996 (Fig. 2.10), suggesting that DOC and TDP leach independently. Therefore, DOC may mitigate leaching of some forms of P, but not others.

Since the water table is maintained by precipitation, TDP concentrations peaked in the summer after rain events that caused the water table to rise and intersect the organic layer. TDP export also increased because both TDP concentrations and water flow (expressed as transmissivities) increased. Although TDP export was higher in the uncut than the cut SC, TDP export coefficients were similar because the uncut SC had a greater area. Estimated TDP export coefficients (14 and 16 $\mu g\ m^{-2}\ d^{-1}$) were similar to mean export coefficient from conifer-dominated (19 $\mu g\ m^{-2}\ d^{-1}$; Munn and Prepas 1986) and from aspen-dominated watersheds (27 $\mu g\ m^{-2}\ d^{-1}$; Cooke and Prepas 1998) on the Boreal Plain. Munn and Prepas (1986) show that export coefficients based on weekly samples would underestimate those based on daily samples. Since the samples were collected on a biweekly basis, export coefficients were likely underestimated. In addition, the period immediately after the storm at the end of July 1997, when TDP export was likely at its highest, was missed. Estimated daily export coefficients of the subsurface flow and streams on the Boreal Plain were within a similar range, suggesting that TDP exports in streams were dominated by subsurface water.

Although the response of TDP concentrations to changes in the water table was similar for both SCs, the lake seemed to influence the response in wells at F1 and G1. In August 1996, TDP concentrations peaked at the same time, but TDP remained high in the uncut SC and not in the cut SC perhaps because flooding occurred in the uncut, but not in

the cut SC (Fig. 2.10). This flooding could have created anoxic conditions, which reduce Fe oxides, releasing bound phosphates (D'Angelo and Reddy 1994). In addition, increased soil water content has been shown to increase TDP concentrations in soil solution (Sharpley 1983). Prolonged flooding of G1 in 1997 may have caused TDP concentrations to peak 2 weeks after F1. The actual peak in TDP concentrations at F1 was probably missed because samples were taken two weeks after the June 1997 storm. The recorded peak concentration was only 96 $\mu g\ L^{-1}$ compared to the peak concentration of 2280 $\mu g\ L^{-1}$ in 1996. However, TDP concentrations at F1 remained much lower after the June storm in 1997 (<189 $\mu g\ L^{-1}$) than the August storm in 1996 (>500 $\mu g\ L^{-1}$). The lake may have washed away some of the organic layer in 1997 when the lake flooded the shore because the shore was open to the lake and subject to wave action, especially in 1997 when the lake level was high (Fig 2.10). Furthermore, waves may have washed P from the soils when the lake was high or may have aerated the soils and reduced P solubility. In contrast, TDP concentrations at G1, which was protected from wave action by a thick stand of willows, had similar ranges in both years. Therefore, the lake may influence TDP concentrations in shallow subsurface water by wave action or flooding of the shore.

Influence of topography

Topography likely influenced TDP and DOC concentrations because nutrients and soil often accumulate in depressional areas, where soil moisture is usually higher (Anderson 1987). The slope in the uncut SC was concave within 100 m of the lake and organic matter had accumulated near the lake at G1 (Fig 2.5). In contrast, organic matter

was less at F1, which is at the base of a convex slope (Fig 2.4). However, organic matter had accumulated at F9, where the slope was concave. The thicker organic layer at G1 (61 cm) provided a greater source of organic P and C than at F1 (25 cm), providing a constant source of P from year to year. Organic C and P may have been flushed more quickly from the thinner organic layer at F1 because convex slopes leach faster than concave slopes (Anderson 1987). Concave slopes retain water for longer than convex slopes, thereby increasing the leaching time and the zone of interaction between runoff and the soil (King et al. 1983; Sharpley and Smith 1989). A longer water residence time may also create anoxic conditions, which may increase P solubility in mineral soils (D'Angelo and Reddy 1994). At F9, water likely flowed from surrounding slopes and accumulated in the depression, increasing the leaching time of P. Since the lithology at F9 was relatively sandy, P could be transported to a deeper depth, as seen with the higher TDP concentration in the deeper piezometer (Fig 2.7). Low clay content and accumulation of P from the surrounding hillslopes likely caused the consistently high TDP concentrations (140 to 265 $\mu g\ L^{-1}$) in the piezometers. Water from this depression likely flowed over the shallow clay layer from F8 to F5, then flowed deeper in the soil from F4 to F1 where the clay was deeper (Fig 2.4). Thus, the concentration from F7 to F2 remained relatively high (>120 $\mu g\ L^{-1}$), even during low flow. TDP concentrations in wells were highest in the areas with the lowest slope (F9 and G1) and likely influenced TDP concentrations in areas downslope from them.

Influence of climate and vegetation

TDP concentrations and exports generally peaked when the water flowed through the organic layer after summer rainstorms probably due to flushing of accumulated P from, and increased flow through, the organic layer. Decomposition rates generally increase over the growing season as soil temperatures rise, when soil moisture is sufficient for microbial activity. Rates slow down in late summer when temperatures decline. Long relatively dry periods allow P to accumulate in the organic layer; the accumulated P can then be flushed from the soil during subsequent rainstorms (Anderson 1988). In both August 1996 and July 1997, TDP concentrations increased after major rain events preceded by relatively dry periods, suggesting that P was flushed from the soils. The magnitude of TDP peaks after subsequent storms were generally not as large perhaps due to a shorter time between storms for P to accumulate. In the spring, the peak in TDP concentrations in the uncut SC was likely due to snowmelt flowing through the organic layer above frozen mineral soils. Water was obviously flowing through the organic layer at the time of sampling. Furthermore, vegetation may influence temporal trends in TDP concentrations because of changes in the demand for phosphates in soil solution by plants (Huang and Schoenau 1997). Therefore, precipitation patterns, climate, and vegetation may influence the magnitude, timing, and duration of elevated TDP concentrations in subsurface flow.

This study began in 1996, an extremely wet year with the second highest annual precipitation in 27 years, thus the water storage capacity was relatively full. In 1997, the year after logging, antecedent soil moisture was high at the beginning of the open-water

season, creating a high potential for water to flush P from the organic layer. Therefore, this study was during a period where the potential for P transport was high.

Influence of forest harvesting

TDP concentrations in piezometers (mineral soil) were not likely influenced by forest harvesting. Several studies observed that TDP concentrations in solution of mineral soils were similar before and after logging probably due to high P adsorption capacity of the soils (e.g. McColl 1978; Yanai 1991). In my study, mean TDP concentrations in piezometers less than 1.7 m deep were not detectably different. However, mean concentrations in piezometers deeper than 1.7 m deep were lower in the cut SC ($P<0.05$). The difference in P concentrations in piezometers is likely due to higher clay content in soils of the cut SC rather than a logging effect.

TDP concentrations in wells were higher in the cutblock than in the uplands of the uncut SC. However, the water table was high in the cutblock and could be the reason for the high TDP concentrations instead of logging (Table 2.4, Fig. 2.9). In this study, differences in mean daily export coefficients were not detectable between the cut and uncut SC. Nonetheless, forest harvesting often increases water yield due to decreased interception and ET and increased soil moisture (Bosch and Hewlett 1982; Keenen and Kimmins 1993) and could have raised the water table in the cut SC. Since the antecedent soil moisture was high in May 1997, the water table was high in the post-harvest year. Thus, differences between the uncut and cut SCs could not be attributed to logging because of differences in the topography and lithology and their influence on the hydrology of the two SCs.

TDP concentrations in wells decreased with distance into the buffer strip when the water table was high and flowed through the organic layer. However, the water table also decreased with distance into the buffer and likely influenced TDP concentrations. When the water table was low and below the organic layer, TDP concentrations were relatively similar at 5 of 6 sites through the buffer strip (Fig 2.7). Although subsurface water flowed through the rooting zone, TDP concentrations increased when water flowed through the organic layer, which includes both the litter and humus layer. In a study in northern Saskatchewan, P uptake by plants was greatest in the humus layer where fine roots were concentrations (Huang and Schoenau 1996). Since TDP concentrations did not decrease when the water table was low, but decreased when the water table fell below the organic layer, the organic layer was likely a greater source than plant uptake was a sink for P.

Although no effects of forest harvesting were detected in this study, logging on the Boreal Plain could still potentially increase P inputs to lakes through P-laden runoff. This study was done during a period of high runoff, which may have obscured any effects of logging. However, if subsurface flow increases after logging, as found in many other studies (Bosch and Hewlett 1982), P export in subsurface water in the boreal mixedwood forest may increase after logging, especially during wet years.

Studies need to be done on P transport from areas of the boreal mixedwood forest with different physiographic features (e.g. concave and convex slopes, lowlands and uplands, wetlands), with and without disturbances, to understand how to minimize P export to lakes and streams. For studies of soil nutrient cycles on the Boreal Plain, multiple-year studies are required to integrate the effect of climate and forest harvesting.

Conclusions

In this study, TDP in shallow subsurface waters had definite spatial and temporal trends. Lower TDP concentrations in deeper subsurface water were likely due to high clay and Ca content in the mineral soils. Higher TDP concentrations occurred in areas where the water table intersected the organic layer. These areas of intersection, and thus flushing of P, increased when the water table rose after summer rainstorms.

Although TDP concentrations were higher in the cut than in the uncut SC, the influence of the different water flow regimes in the two SCs made it difficult to separate site differences from effects of logging. Estimated TDP export coefficients were not detectably different between the cut and uncut SCs. The buffer strip did not appear to reduce TDP concentrations and exports; however, the organic layer in the buffer was likely a greater source than plant uptake was a sink for P in shallow subsurface flow. Therefore, if water yield increases after logging on the Boreal Plain, the potential for P export to lakes will increase, especially from low-lying areas, where the water table is naturally close to the surface. Further studies should consider the relationship between P dynamics and physiographic characteristics of the boreal mixedwood landscape when evaluating the implication of forest harvesting on the Boreal Plain.

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Table 2.1: General characteristics of the cut and uncut subcatchments

Characteristic	cut	uncut
Area (ha)	2.7	5.1
Mean slope (%) along transect (draw)	11	8
Range of slope (%) to topographic divide (distance to divide: 20 to 140 m)	10 to 20	15 to 30
Aspect	WSW	WNW
% conifer along center transect[a]	39	33
% deciduous along center transect[a]	61	67

[a] for seven plots, each with an area of 9 m^2

Table 2.2: Range and mean hydraulic conductivities (K_{sat}) for mineral soils in both subcatchments compared to values for similar soils in the literature [a]

Subcatchments K_{sat} (m d^{-1})			Range from literature K_{sat} (m d^{-1})	
Soil type	Geometric mean	Range	Soil type	Range
Sand	1.75×10^{0}	1.23×10^{0} to 2.95×10^{0}	Clean sand	1.72×10^{-1} to 8.64×10^{-2}
Silty sand	9.20×10^{-2}	1.03×10^{-2} to 9.76×10^{-1}	Silty sand	6.86×10^{-3} to 6.86×10^{1}
Sandy clay	1.07×10^{-3}	1.44×10^{-4} to 8.54×10^{-3}	Glacial till	6.86×10^{-8} to 1.09×10^{-1}
Clay	7.72×10^{-6}	9.00×10^{-8} to 9.24×10^{-5}	Clay	3.44×10^{-8} to 1.37×10^{-4}

[a] adapted from Freeze and Cherry 1979

Table 2.3: Mean transmissivities with and without saturation of the organic layer for wells in both subcatchments

Subcatchment	Transmissivity ($m^2\ d^{-1}$) with saturated organic layer			Transmissivity ($m^2\ d^{-1}$) without saturated organic layer		
	n	Mean (SE)	Range	n	Mean (SE)	Range
Cut	47	1.7×10^{1} (1.3×10^{0})	2.3×10^{0} to 3.6×10^{1}	60	2.1×10^{-1} (4.1×10^{-2})	1.1×10^{-6} to 1.2×10^{0}
Uncut	70	1.2×10^{1} (2.0×10^{0})	8.0×10^{-1} to 7.1×10^{1}	81	6.2×10^{-2} (5.0×10^{-2})	1.3×10^{-7} to 1.4×10^{0}
Cut and Uncut	117	1.4×10^{1} (1.3×10^{0})	8.0×10^{-1} to 7.1×10^{1}	141	1.1×10^{-1} (3.3×10^{-2})	1.3×10^{-7} to 1.4×10^{0}

Note: Transmissivities were calculated for 13 wells from the cut and 15 wells from the uncut subcatchment for each sampling date with the mean K_{sats} in Table 2.2 and a K_{sat} of 100 m d^{-1} for the organic layer.

Table 2.4: Spearman correlation coefficients for TDP and selected parameters in wells of cut (below diagnol) and uncut (above diagnol) subcatchments in 1997. Number of pairs is in brackets below each correlation.

Parameter[a]	TDP	DOC	Depth	Trans	D.O.	Temp	pH	Cond	Ca	K	Na
TDP		0.57*** (165)	-0.58*** (175)	0.67*** (132)	-0.25* (152)	0.26* (153)	ns (175)	ns (175)	0.49*** (94)	0.44*** (94)	0.51*** (94)
DOC	0.74*** (127)		-0.43*** (165)	0.50*** (125)	ns (143)	ns (143)	-0.46*** (165)	0.32*** (165)	ns (94)	0.62*** (94)	0.31* (94)
Depth	-0.62*** (130)	0.44*** (128)		-0.66*** (151)	0.46*** (156)	ns (157)	0.40*** (175)	-0.39*** (193)	-0.65** (94)	-0.41*** (94)	-0.62*** (94)
Trans	0.72*** (93)	0.62*** (93)	-0.88*** (93)		-0.38*** (119)	ns (120)	-0.32** (132)	0.31*** (146)	0.44** (94)	ns (94)	0.46*** (94)
D.O.	ns (113)	-0.30* (111)	ns (115)	ns (93)		-0.29*** (156)	ns (152)	-0.41*** (152)	-0.61*** (94)	ns (94)	-0.51*** (94)
Temp	ns (113)	0.31* (111)	ns (115)	ns (93)	-0.30* (115)		ns (153)	ns (153)	ns (94)	ns (94)	ns (94)
pH	ns (130)	ns (128)	ns (131)	ns (93)	ns (114)	ns (114)		-0.44*** (175)	-0.53*** (94)	ns (94)	-0.45*** (94)
Cond	-0.51*** (130)	ns (128)	ns (148)	-0.36** (93)	ns (114)	ns (114)	ns (131)		0.99*** (94)	ns (94)	0.88*** (94)
Ca	-0.41** (85)	ns (85)	ns (85)	-0.36* (85)	ns (85)	ns (85)	ns (85)	0.93*** (85)		ns (94)	0.88*** (94)
K	0.86*** (85)	0.52*** (85)	ns (85)	ns (85)	ns (85)	ns (85)	ns (85)	-0.51*** (85)	-0.47*** (85)		0.30* (94)
Na	-0.54*** (85)	-0.32* (85)	ns (85)	ns (85)	ns (85)	ns (85)	ns (85)	0.77*** (85)	0.64*** (85)	-0.66*** (85)	

*,**,*** correlation is significant at the 0.05, 0.01, and 0.001 level, respectively (Bonferroni adjusted significance levels of 0.0045, 0.0009, 0.00009, respectively)

[a] depth, water table depth; trans, transmissivity; D.O., dissolved oxygen; temp, temperature; Cond, specific conductance

Table 2.5: Spearman correlation coefficients for TDP and selected parameters in piezometers of cut (below diagnol) and uncut (above diagnol) subcatchments, 1997. Number of pairs is in brackets below each correlation.

Parameter	TDP	DOC	Pzdepth	Head	K_{sat}	pH	Cond	Ca	K	Na
TDP		**0.48**** (60)	**-0.32***** (224)	**ns** (223)	**0.23*** (215)	**ns** (223)	**-0.29***** (223)	**-0.75**** (19)	**ns** (19)	**-0.84***** (19)
DOC	**0.40**** (79)		**-0.54***** (60)	**ns** (60)	**ns** (60)	**ns** (60)	**-0.57***** (60)	**ns** (19)	**ns** (19)	**ns** (19)
Pzdepth	**-0.58***** (257)	**-0.54***** (79)		**ns** (397)	**-0.28***** (362)	**ns** (280)	**ns** (324)	**ns** (19)	**ns** (19)	**ns** (19)
Head	**0.28***** (254)	**ns** (79)	**ns** (409)		**0.24***** (356)	**ns** (278)	**-0.53***** (322)	**-0.62*** (19)	**ns** (19)	**ns** (19)
K_{sat}	**0.28***** (253)	**ns** (78)	**ns** (404)	**0.31***** (399)		**0.26***** (264)	**-0.56***** (308)	**ns** (19)	**ns** (19)	**ns** (19)
pH	**ns** (256)	**ns** (79)	**ns** (314)	**ns** (309)	**ns** (309)		**-0.51***** (280)	**-0.83***** (19)	**ns** (19)	**-0.87***** (19)
Cond	**-0.78***** (256)	**-0.40**** (79)	**ns** (365)	**-0.23***** (360)	**-0.37***** (358)	**ns** (311)		**1.00***** (19)	**ns** (19)	**0.93***** (19)
Ca	**-0.89***** (20)	**ns** (20)	**ns** (20)	**ns** (20)	**ns** (19)	**ns** (20)	**0.99***** (20)		**ns** (19)	**0.94***** (19)
K	**0.65*** (20)	**0.61*** (20)	**ns** (20)	**ns** (20)	**ns** (19)	**ns** (20)	**-0.62*** (20)	**ns** (20)		**ns** (19)
Na	**-0.83***** (20)	**ns** (20)	**ns** (20)	**ns** (20)	**ns** (19)	**ns** (20)	**0.90***** (20)	**0.88***** (20)	**ns** (20)	

*,**,*** correlation is significant at the 0.05, 0.01, and 0.001 level, respectively (Bonferroni adjusted significance levels of 0.0045, 0.0009, 0.00009, respec

[a] Pzdepth, depth of piezometer screening; head, pressure head; Cond, specific conductance

Table 2.6: Estimated daily P export and export coefficient in the cut and uncut subcatchments

Date	Cut		Uncut	
	export (mg)	coefficient ($\mu g\ m^{-2}$)	export (mg)	coefficient ($\mu g\ m^{-2}$)
06/18/97	263	10	1588	31
07/15/97	563	21	6933	135
07/29/97	509	19	2403	47
08/12/97	142	5	7	0.1
08/24/97	255	9	702	14
09/09/97	2908	11	1185	23
09/23/97	350	13	705	14
Mean	431	16	722	14

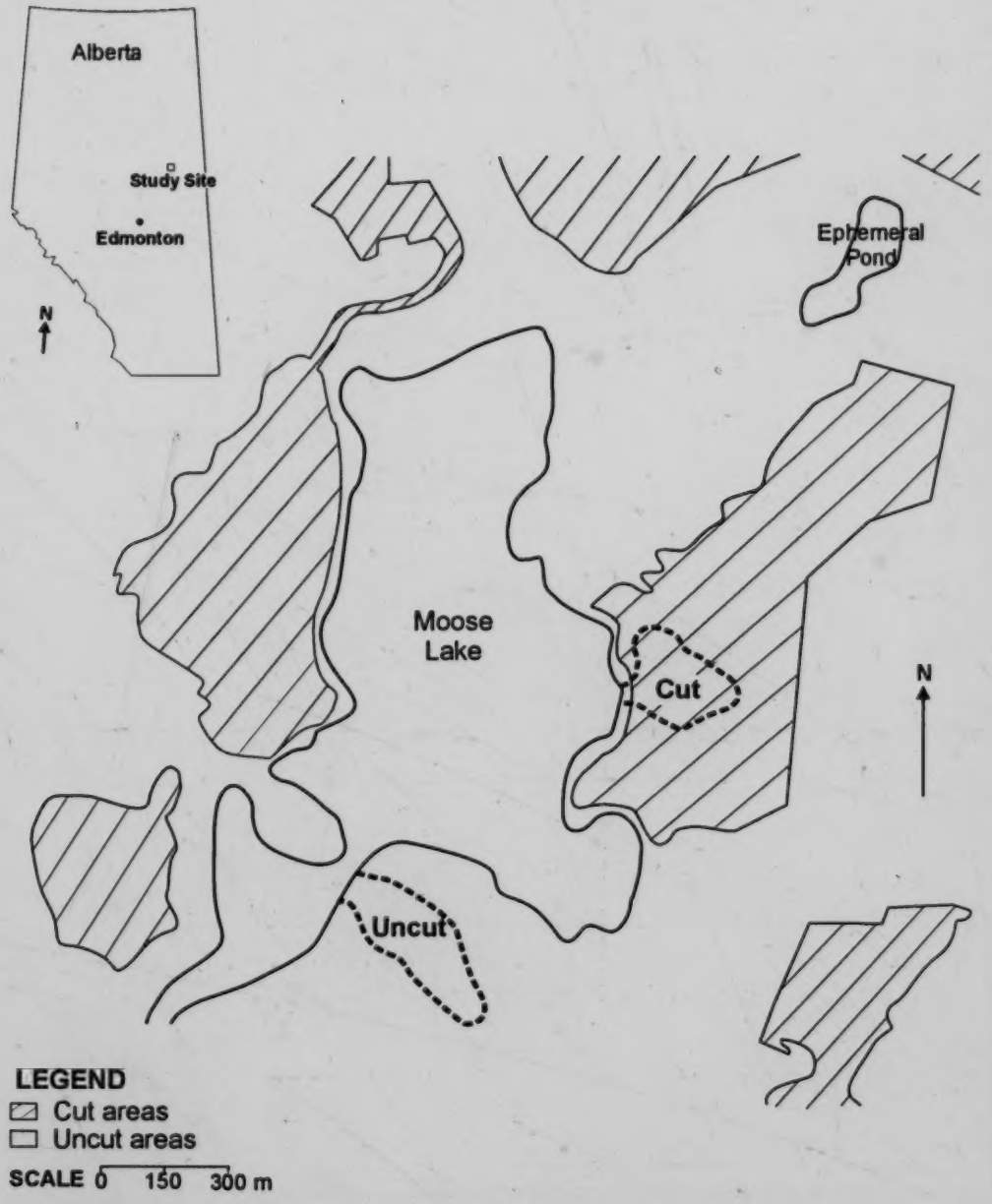
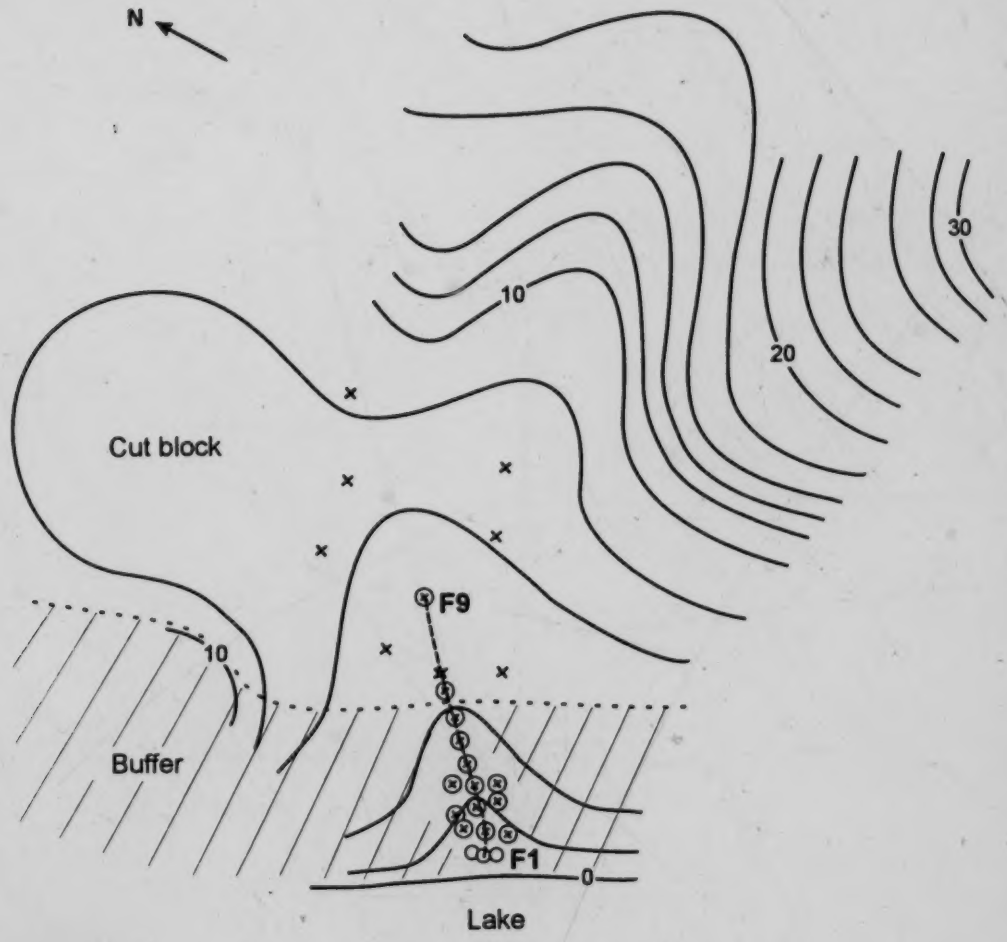


Figure 2.1: Location of subcatchments and distribution of cut blocks at Moose Lake.



LEGEND
- ○ Single well
- × Single piezometer
- ⊗ Nested wells and piezometers

SCALE 0 20 40

Figure 2.2: Instrument sites and topography of cut subcatchment with a treed buffer. The line ending at F1 and F9 represents the center transect. Contour intervals = 2 m. Elevation is meters above lake level on October 10, 1997.

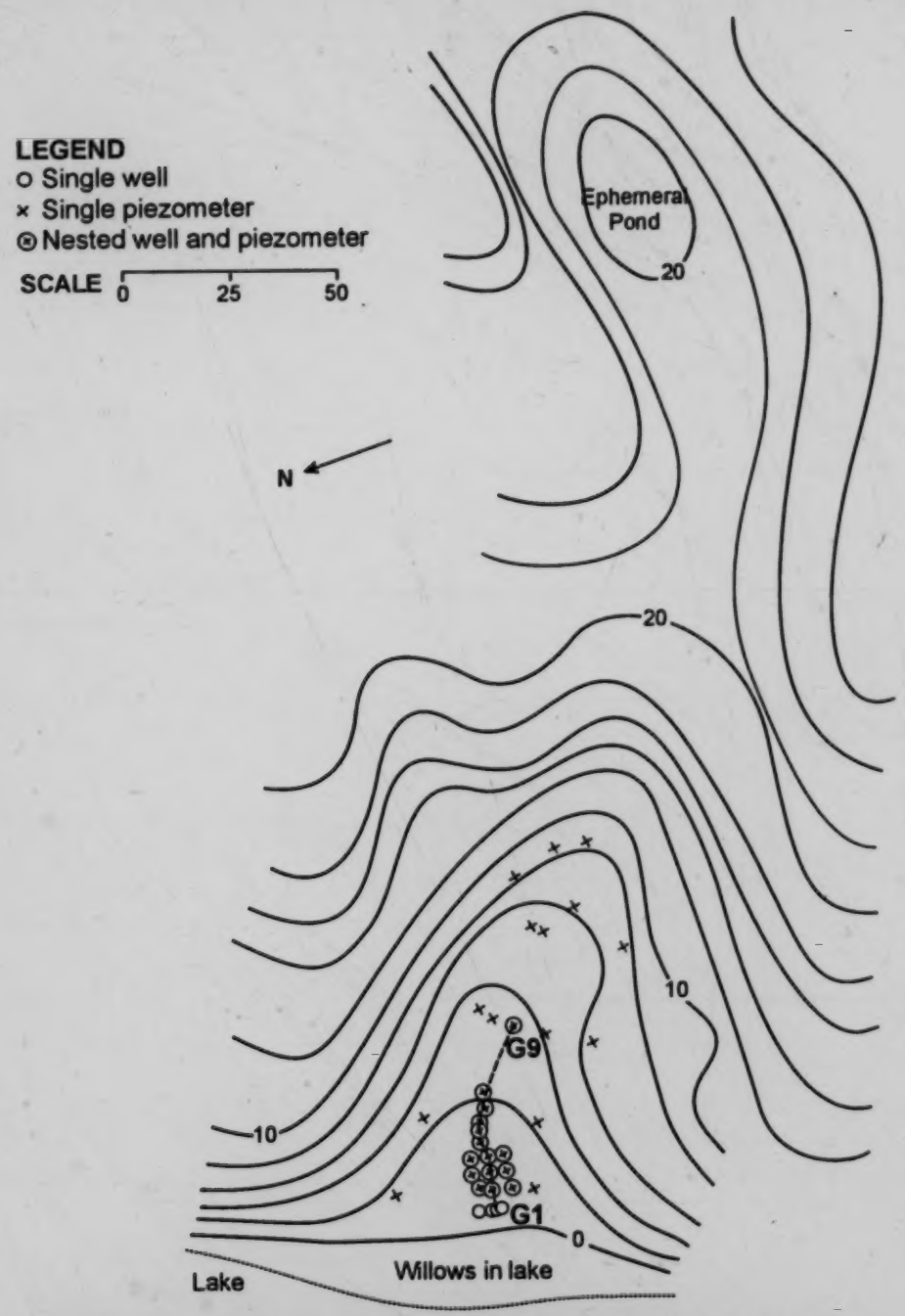


Figure 2.3: Well and piezometer sites and topography of uncut subcatchment. Line ending at G1 and G9 represents the center transect. Contour intervals = 2 m. Elevation is meters above lake level on October 10, 1997.

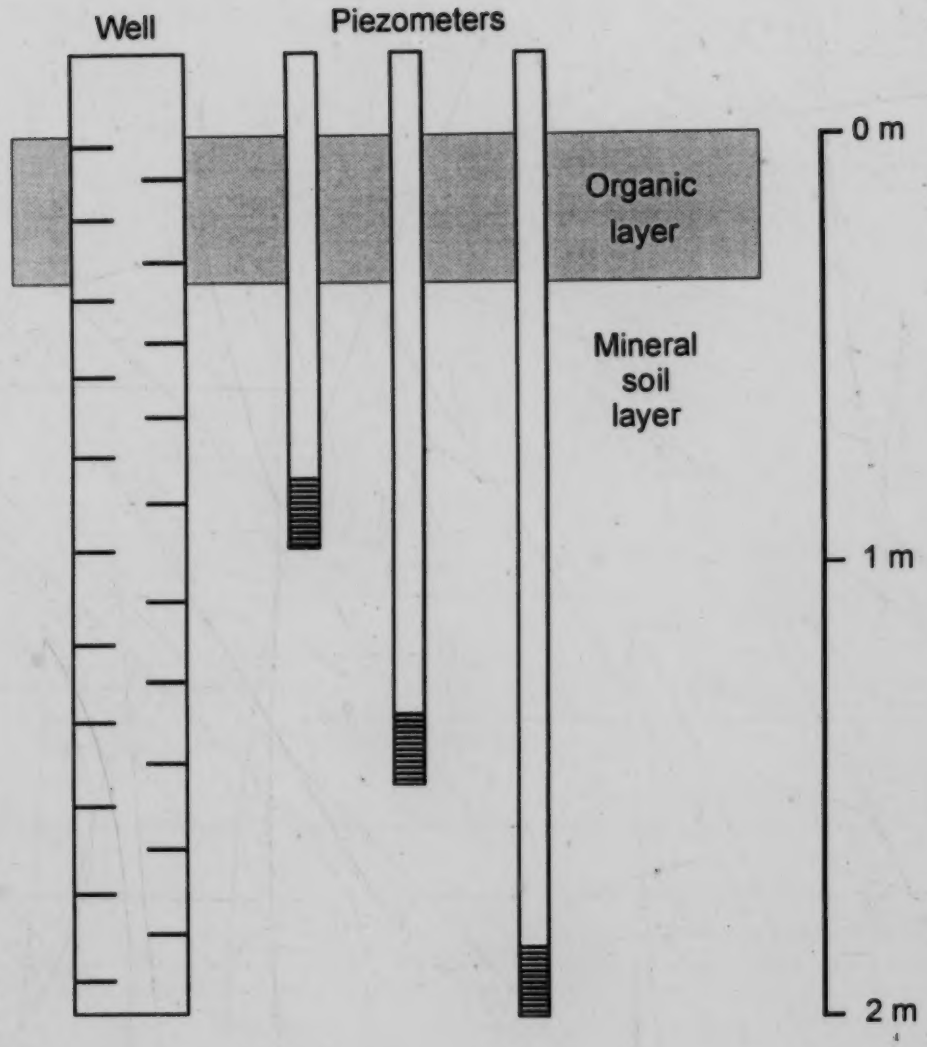


Figure 2.4: Schematic of nested piezometers and wells

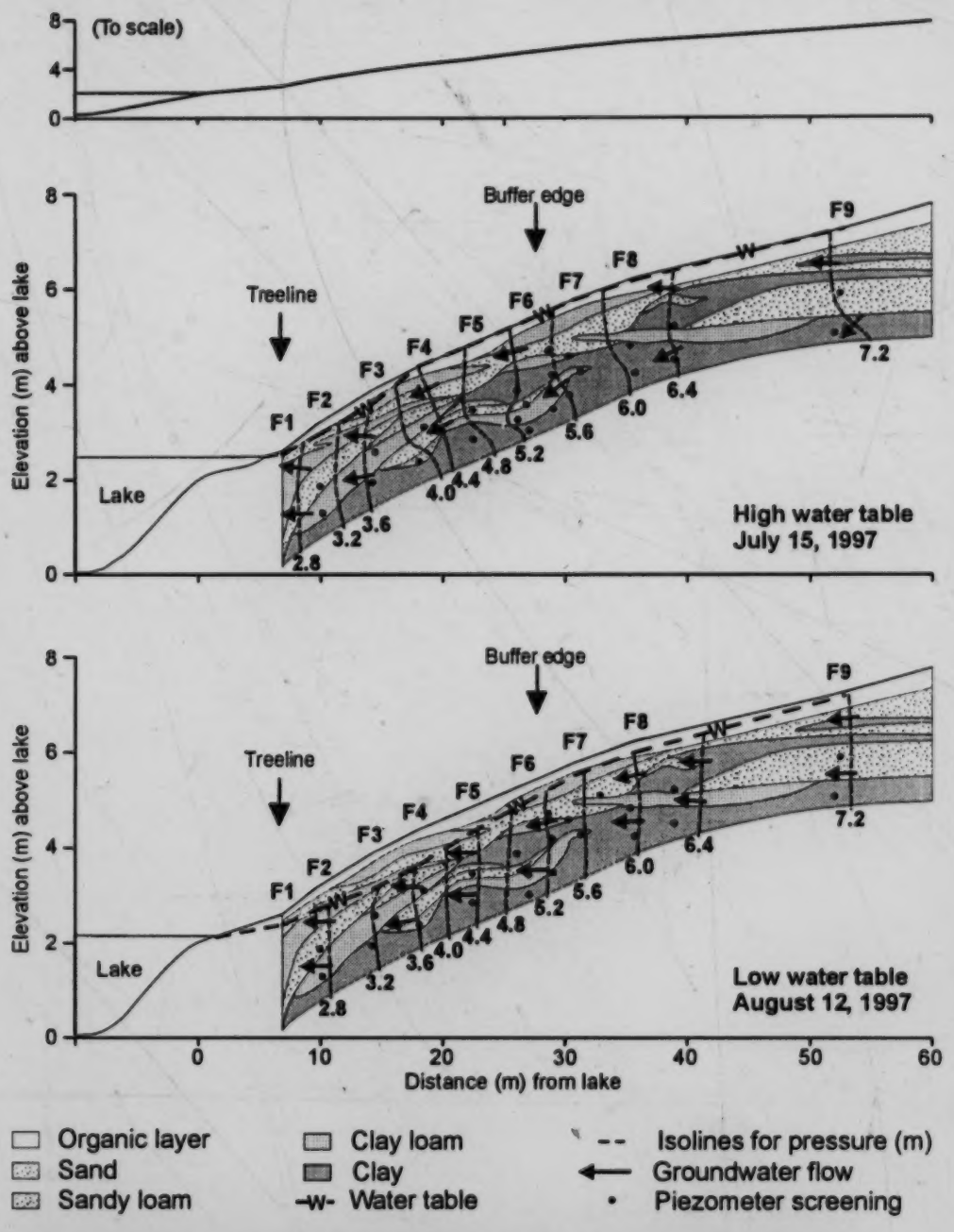


Figure 2.5: Lithology, flow net, and subsurface flow along the center transect of cut subcatchment when the water table was a) high and b) low in 1997. Sites are shown in Fig. 2.2. Numbers are head or level in m. Note lithology was interpolated between sites.

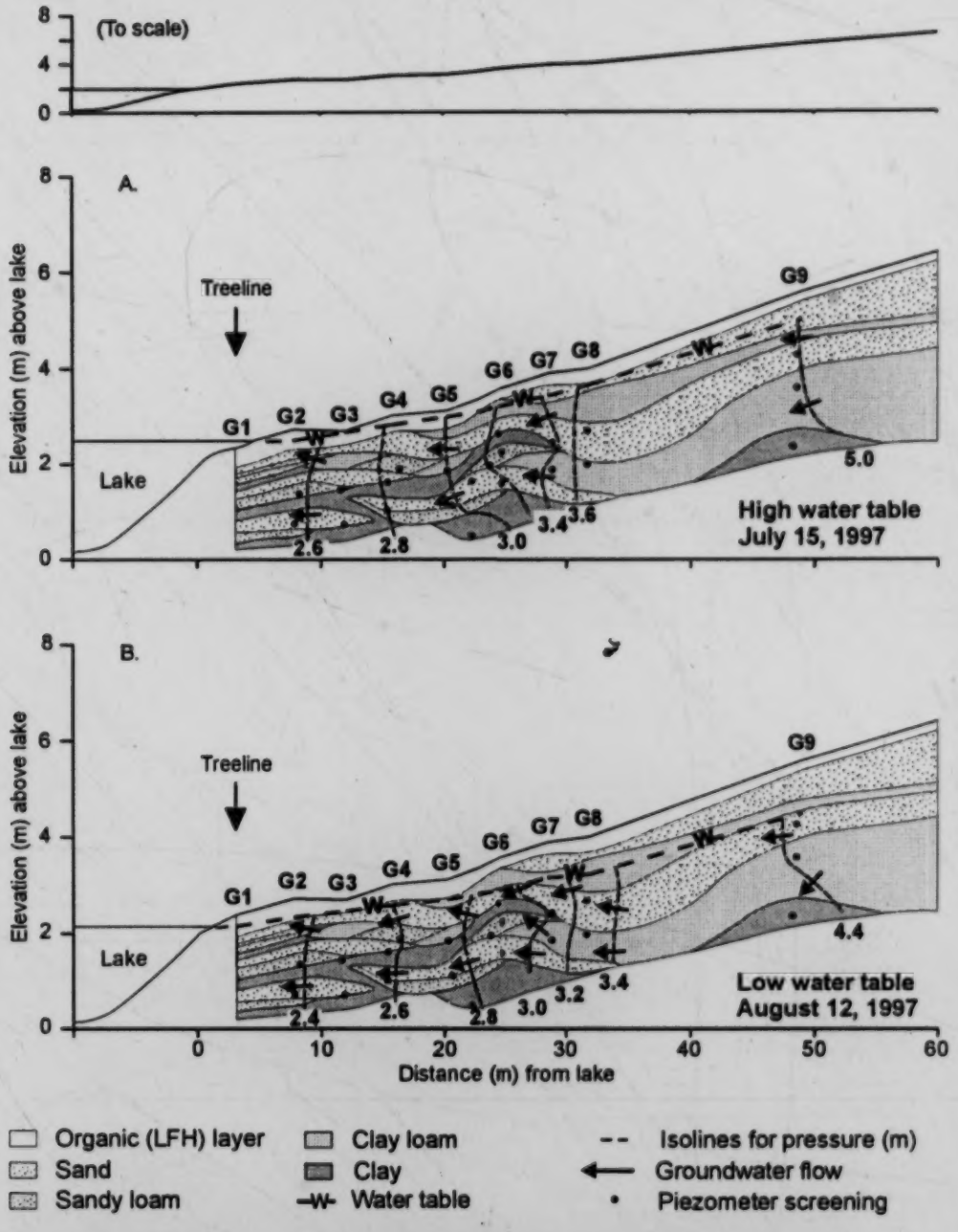


Figure 2.6: Lithology, flow net, and subsurface flow along the center transect of the uncut subcatchment when the water table was a) high and b) low in 1997. Sites are shown in Fig. 2.3. Numbers are head or level in m. Note that lithology was interpolated between sites.

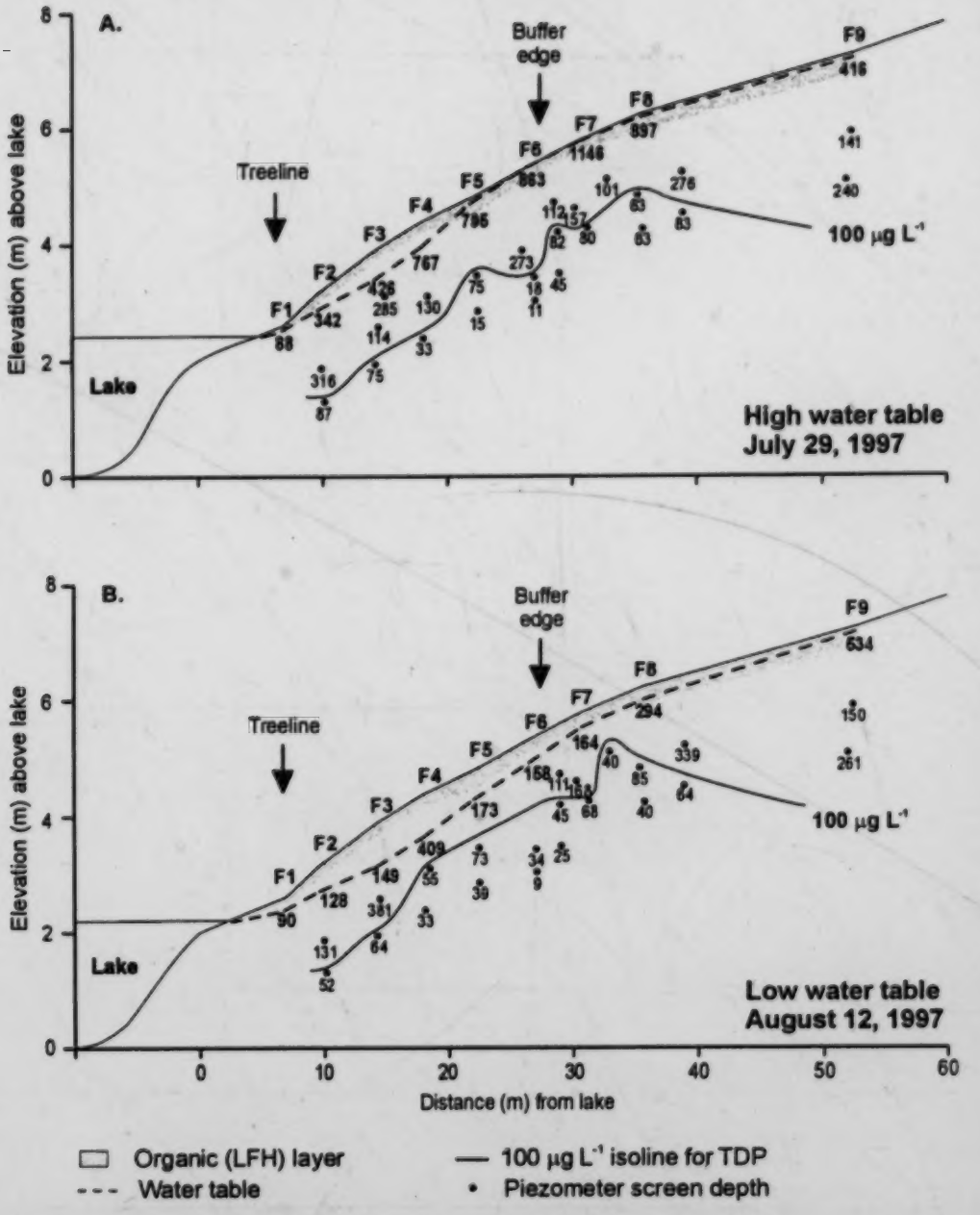


Figure 2.7: TDP concentrations (µg L^{-1}) in piezometers (•) and wells (bold numbers below water table) along the center transect of the cut SC when the water table was a) high and b) low in 1997. Isolines separate concentrations above and below 100 µg L^{-1}.

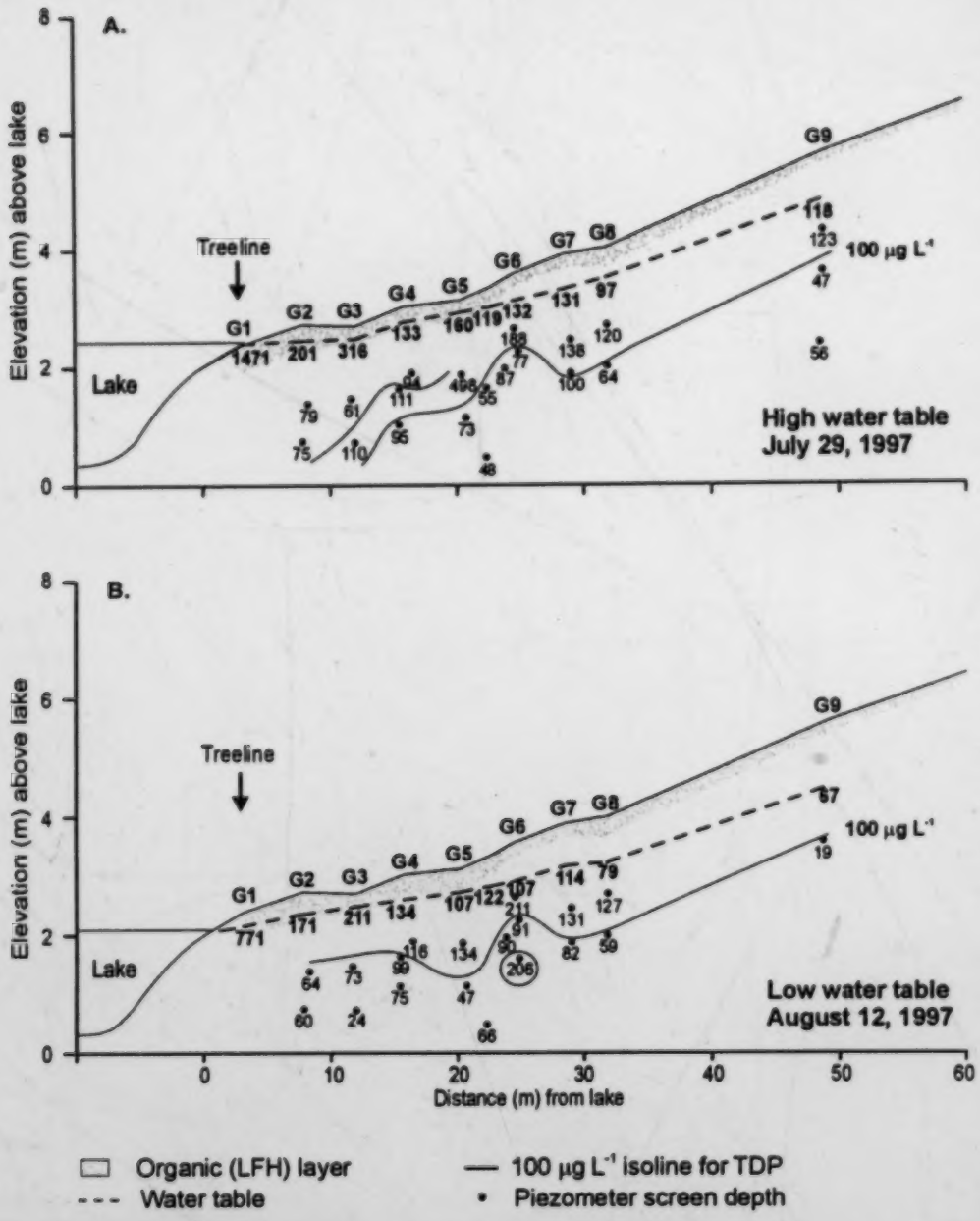


Figure 2.8: TDP concentrations (µg L^{-1}) in piezometers (•) and wells (bold numbers below water table) along the center transect of the uncut SC when the water table was a) high and b) low in 1997. Isolines separate concentrations above and below 100 µg L^{-1}.

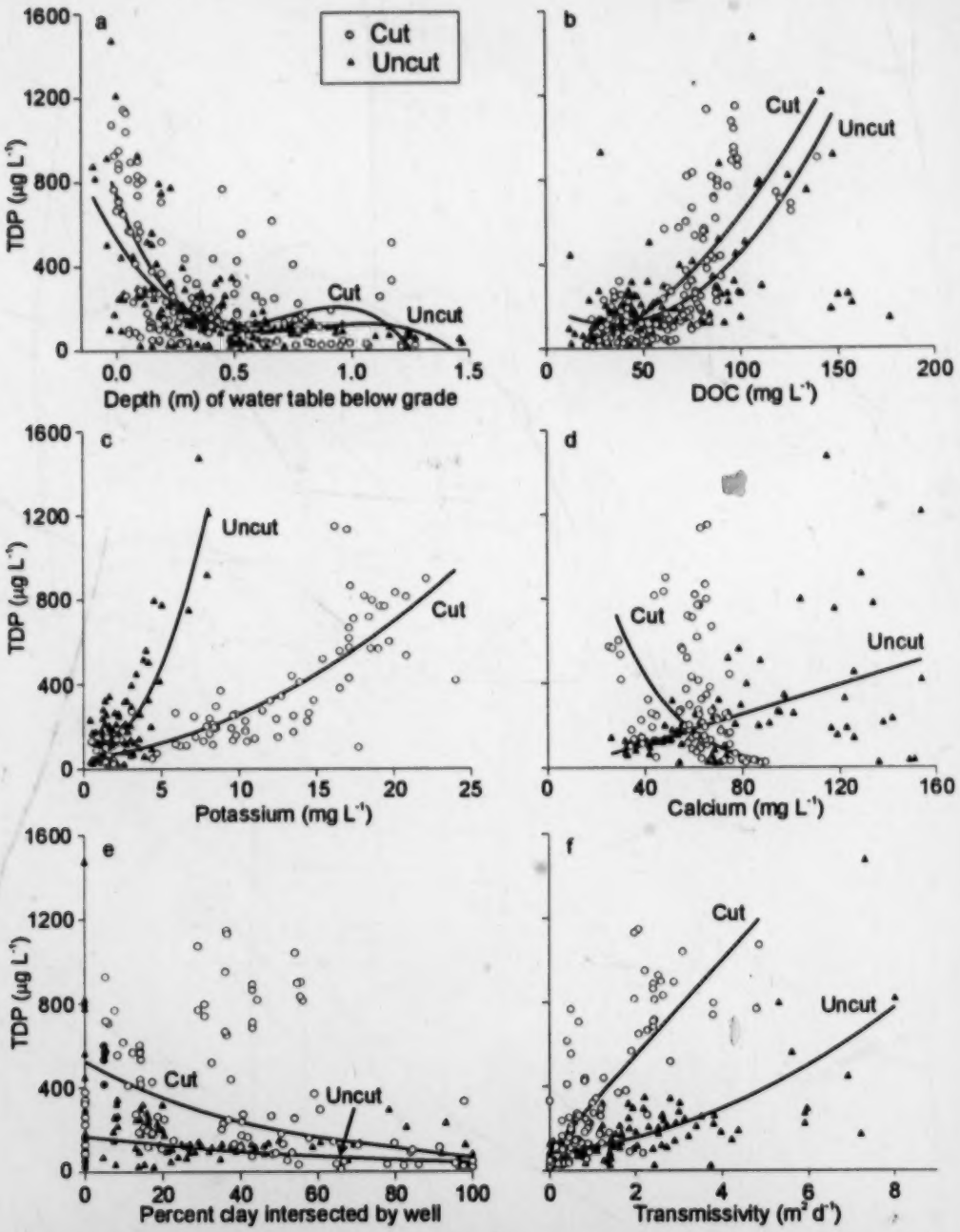


Figure 2.9: Relationships between TDP concentrations in wells and a) depth of water table below ground, b) dissolved organic carbon (DOC), c) potassium, d) calcium, e) percent saturated clay intersected by wells, and f) transmissivity. R-squared values and regression equations are in Table A1 (Appendix).

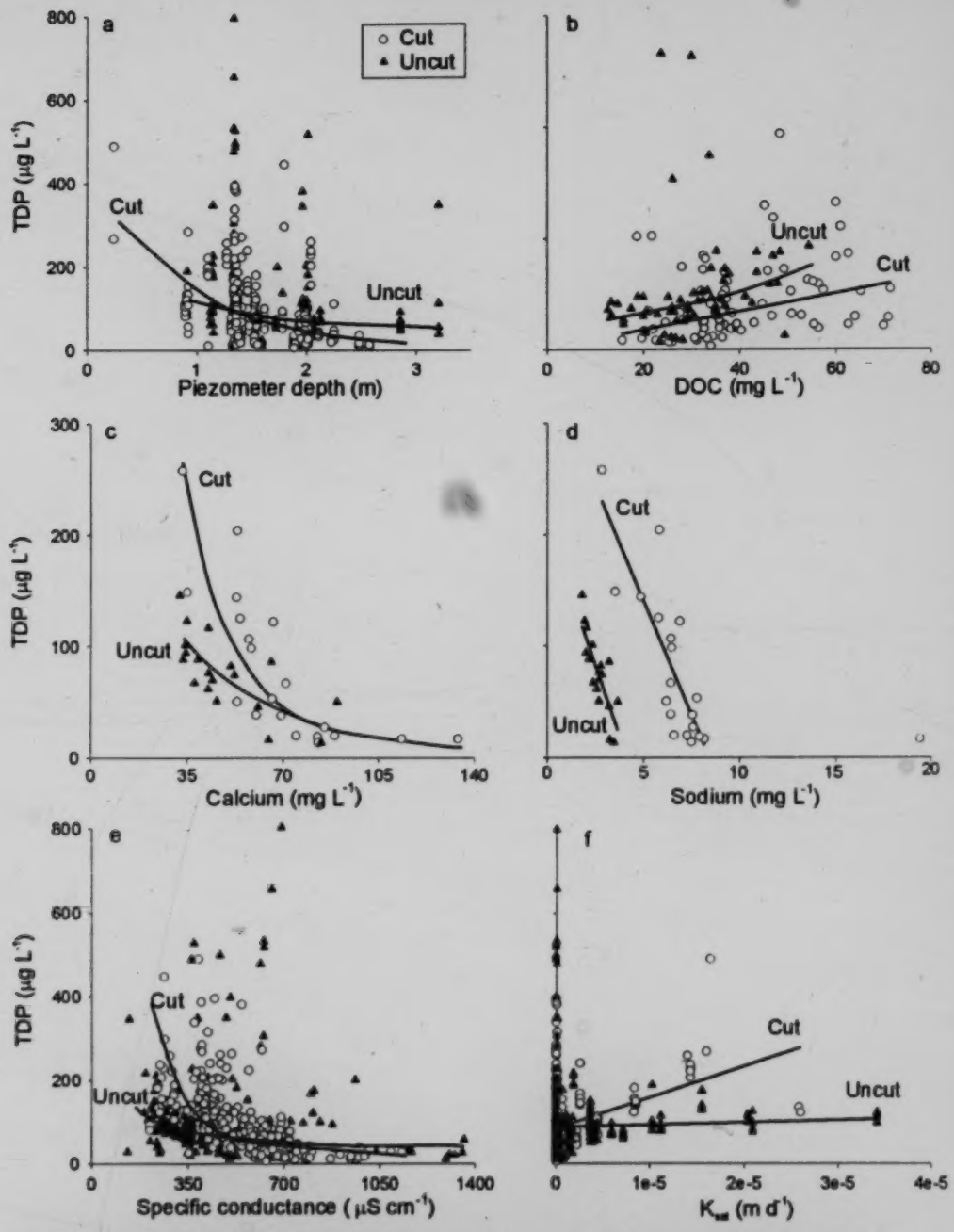


Figure 2.10: Relationships between TDP concentrations in piezometer and a) piezometer depth, b) dissolved organic carbon, c) calcium, d) sodium, e) specific conductance and f) hydraulic conductivity (K_{sat}). R-squared values and regression equations are in Table A1 (Appendix)..

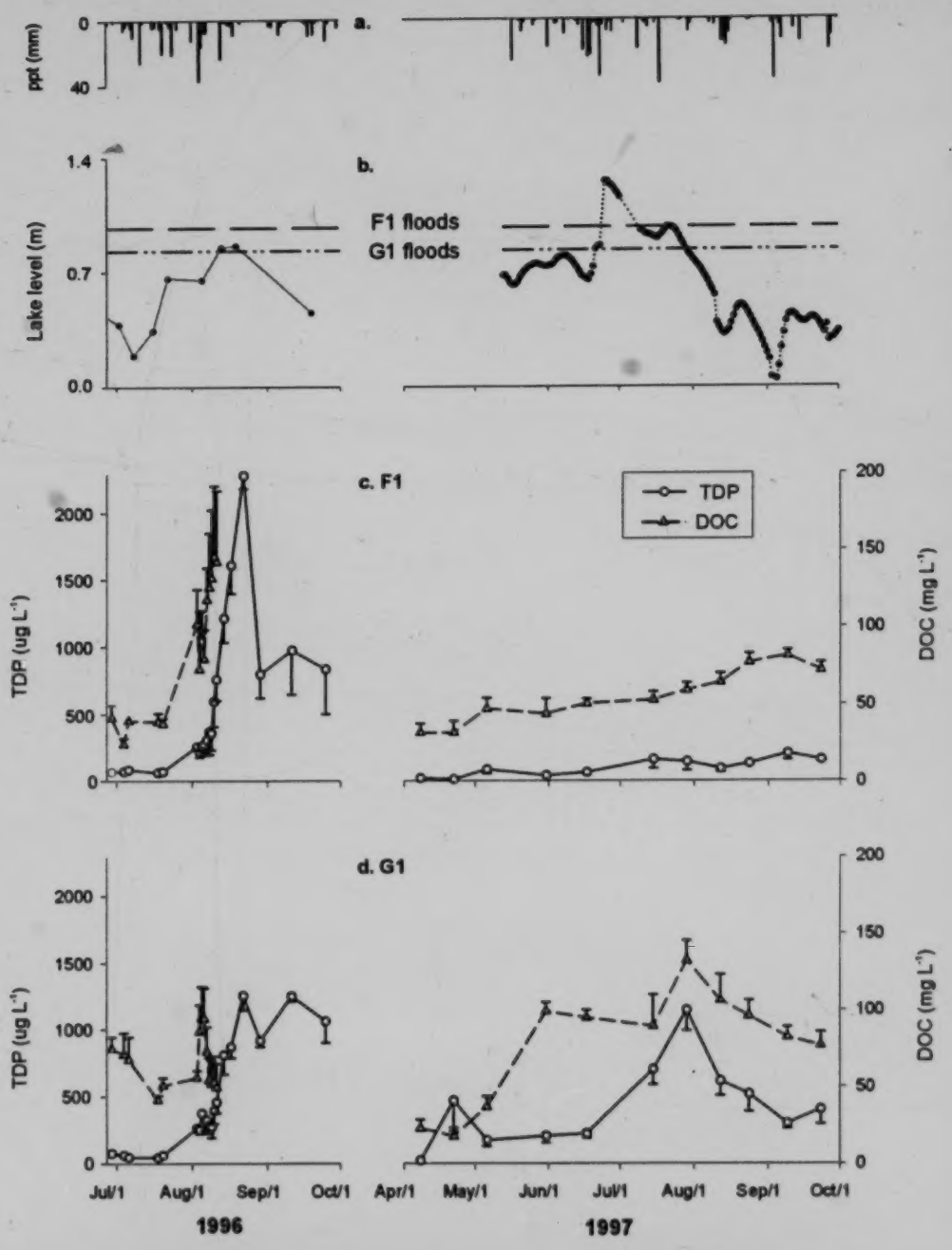


Figure 2.11: a) Precipitation and b) lake levels (above arbitrary datum) for Moose Lake. Mean TDP and DOC concentrations (with SE bars) for the three wells closest to the lake at c) F1 (cut) and d) G1 (uncut) for 1996 and 1997.

Appendix

Table A1: Equations and R-squared values for regression lines between TDP and selected parameters (shown in Fig. 2.9 and 2.10) for wells and piezometers

Parameter	Subcatchment	n	Equation	r^2
Wells				
Water table depth	cut	130	$y = -2119.3x^3 + 4581.3x^2 - 3006.3x + 744.3$	0.42
	uncut	175	$y = -887.5x^3 + 2237.8x^2 - 1758.8x + 533.8$	0.48
DOC	cut	127	$y = 0.0488x^2 + 2.04x - 62.9$	0.54
	uncut	165	$y = 0.0763x^2 - 5.16x + 204.8$	0.53
Potassium	cut	85	$y = 22.0x^2 - 42.1x + 144.1$	0.73
	uncut	94	$y = 1.16x^2 + 9.22x + 47.6$	0.68
Calcium	cut	85	$y = 1377.7e^{-0.0318x}$	0.21
	uncut	94	$y = 3.44x - 24.3$	0.24
Percent clay	cut	95	$y = 542.6e^{-0.0208x}$	0.27
	uncut	136	$y = 164.2e^{-0.0118x}$	0.21
Transmissivity	cut	95	$y = 227.1x + 91.9$	0.58
	uncut	136	$y = 8.34x^2 + 19.5x + 84.5$	0.52
Piezometers				
Piezometer depth	cut	361	$y = 433.6e^{-1.1606x}$	0.30
	uncut	388	$y = 115.07x^{-0.7411}$	0.06
DOC	cut	79	$y = 1.7629x^{0.9796}$	0.16
	uncut	60	$y = 41.669e^{0.0227x}$	0.10
Calcium	cut	20	$y = 964015x^{-2.337}$	0.71
	uncut	19	$y = -79.4\ln(x) + 382.0$	0.54
Sodium	cut	20	$y = -41.1x + 346.6$	0.70
	uncut	19	$y = -40.0x + 208.0$	0.67
Specific conductance	cut	256	$y = 7x10^{-6}x^{-1.848}$	0.50
	uncut	223	$y = 2714x^{-0.604}$	0.14
K_{sat}	cut	353	$y = 8x10^{6}x + 83.038$	0.12
	uncut	315	$y = 147.02x^{0.0414}$	0.03